

### III. STATUS OF AFFECTED SPECIES AND ENVIRONMENTAL BASELINE

The following endangered and threatened species occur in the action area and may be affected by the continued regulation of domestic fisheries in the Western Pacific Region under the Pelagics FMP:

<b>Marine Mammals</b>	<b>Status</b>
Blue whale ( <i>Balaenoptera musculus</i> )	Endangered
Fin whale ( <i>Balaenoptera physalus</i> )	Endangered
Hawaiian monk seal ( <i>Monachus schauinslandi</i> )	Endangered
Humpback whale ( <i>Megaptera novaeangliae</i> )	Endangered
Northern right whale ( <i>Eubalaena glacialis</i> )	Endangered
Sei whale ( <i>Balaenoptera borealis</i> )	Endangered
Sperm whale ( <i>Physeter macrocephalus</i> )	Endangered
 <b>Sea turtles</b>	 <b>Status</b>
Green turtle ( <i>Chelonia mydas</i> )	Endangered/Threatened
Hawksbill turtle ( <i>Eretmochelys imbricata</i> )	Endangered
Leatherback turtle ( <i>Dermochelys coriacea</i> )	Endangered
Loggerhead turtle ( <i>Caretta caretta</i> )	Threatened
Olive ridley turtle ( <i>Lepidochelys olivacea</i> )	Endangered/Threatened

Except for the Hawaiian monk seal, critical habitat for all of the above-listed species in the Pacific Ocean has not been designated or proposed within the action area. In May 1988, NMFS designated critical habitat for the Hawaiian monk seal out from shore to 20 fathoms in 10 areas of the Northwestern Hawaiian Islands. Critical habitat for these species includes “all beach areas, sand spits and islets, including all beach crest vegetation to its deepest extent inland, lagoon waters, inner reef waters, and ocean waters out to a depth of 20 fathoms around the following: Kure Atoll, Midway Islands, except Sand Island and its harbor, Lisianski Island, Laysan Island, Maro Reef, Gardner Pinnacles, French Frigate Shoals, Necker Island, and Nihoa Island” (50 CFR § 226.201). Some U.S. fisheries regulated under the Pelagics FMP fish in critical habitat areas of the Hawaiian monk seal (i.e., ocean waters out to 20 fathoms depth), although they do not adversely affect physical features identified as critical habitat. In addition, these fisheries do not target or incidentally catch prey species of the Hawaiian monk seals. Therefore, the proposed action is not likely to adversely affect critical habitat of the Hawaiian monk seal.

Although blue whales, fin whales, northern right whales, and sei whales are found within the action area and could potentially interact with the U.S. fisheries under the Pelagics FMP, there have been no reported or observed incidental takes of these species in these fisheries. Therefore, the proposed action is not likely to adversely affect blue whales, fin whales, northern right whales, or sei whales, and these species will not be considered further in this Opinion.

In 1991, one humpback was reported by an observer entangled in the mainline of a Hawaii-based longline vessel. The animal was released with trailing gear (Dollar, 1991). The interaction occurred

inside what is now the protected species zone (50 nautical miles) of the islands and atolls of the Northwestern Hawaiian Islands (Bob Harman, NMFS, personal communication, November, 2000). Another humpback whale was reported entangled in longline gear off Lanai (Nitta and Henderson, 1993) and by whalewatch operators off Maui in 1993 (Hill and DeMaster, 1999). Confirmation was not made as to whether the gear type was pelagic longline gear, and it is believed to be the same whale.

Humpback whales favor waters less than 100 fathoms (183 meters) around the main Hawaiian Islands. The highest densities of humpback whales occur in the shallow-water, inter-island channels of the four-island region (Maui, Lānaʻi, Molokaʻi, and Kahoʻolawe) and Penguin Bank (Hudnall, 1978, Baker and Herman, 1981, Mobley and Bauer, 1991 *in* Mazzuca *et al.*, 1998). Because humpback whales prefer shallower waters and the 1991 interaction occurred inside the 50 nautical mile area now closed to longline fishing, NMFS considers the likelihood of another interaction low and does not expect the Hawaii-based longline fishery to interact with a humpback whale. In addition, there have been no reported interactions between humpbacks and other fisheries under the Pelagics FMP. Therefore, NMFS has determined that the proposed action is not likely to adversely affect humpback whales.

NMFS has observed one sperm whale interaction by the Hawaii-based longline fishery. The event occurred in May, 1999 inside the Northwestern Hawaiian Islands EEZ (about 140 nautical miles north of Raita Bank), and the vessel was targeting swordfish (gear was set at night, lightsticks were used, and no line shooter was used). According to the observer report, the sperm whale's pectoral fin was entangled in the mainline. The captain stopped the boat, let out more mainline, and then backed up until he could reach the other end of the mainline. At this point, both ends of the mainline, on each side of the sperm whale, were secured on the vessel. During this time, the whale broke the mainline and swam away without trailing gear. This is the first reported interaction by the observer program since the Hawaii-based longline fleet has been monitored (1991). In addition, there have been no reported sperm whale interactions by fishers in their logbook submissions.

NMFS has observed 3,251 sets, representing approximately 3,874,635 hooks (data from February 1994 through December 31, 1999), since the implementation of the mandatory observer program. Based on this information, the observed entanglement rate for sperm whales would equal approximately 0.31 whales per 1,000 sets or 0.0002 per 1,000 hooks. However, with only one sperm whale entanglement, NMFS believes that this estimated entanglement rate does not represent the actual entanglement rate. One whale entanglement cannot provide a reliable estimate of the true entanglement rate with any certainty. At this time, there is insufficient data to suggest that a sperm whale interaction with longline gear is anything more than a one time random event. Nevertheless, NMFS recognizes the potential that sperm whales could interact with longline gear set in the open water but without more accurate data is unable to predict with any level of confidence the likelihood of an interaction. Therefore, without additional information to support the frequency of entanglements, NMFS does not anticipate that there will be another sperm whale interaction in the foreseeable future by the Hawaii-based longline fishery. In addition, there have been no reported interactions between sperm whales and other fisheries under the Pelagics FMP. Therefore, NMFS has determined that the proposed action is not likely to adversely affect sperm whales.

The endangered Hawaiian monk seal is currently found throughout the NWHI, specifically: Kure Atoll, Midway Islands, Pearl and Hermes Reef, Lisianski Island, Laysan Island, French Frigate Shoals, Gardner Pinnacles, Necker Island and Nihoa Island. These islands form a chain approximately 1,840 km long. Hawaiian monk seals are also occasionally found in the main Hawaiian Islands. The longline area closure around the NWHI instituted in 1991 (longline fishing prohibited within 50 nm of the NWHI and in 100 nm closed corridors connecting the non-contiguous closed circles) appears to have eliminated monk seal interactions with the Hawaii-based longline fleet, as there have been no observed or reported interactions with this fishery since then. In addition, there have been no reported interactions between Hawaiian monk seals and other fisheries under the Pelagics FMP. Therefore, NMFS has determined that the proposed action is not likely to adversely affect the Hawaiian monk seal.

Although hawksbill turtles are known to nest on the Main Hawaiian Islands (Molokai and Hawaii), they are not known to interact with the Hawaii-based longline fishery, as there have been no reported or observed interactions between these pelagic longliners and hawksbills. As hawksbills become adults, evidence suggests that they switch foraging behaviors from pelagic surface feeding to benthic reef feeding. The maturing turtle establishes foraging territory and will remain in this territory until it is displaced. If Hawaiian hawksbills forage close to their known nesting sites, they are probably benefitting from the protected species zone instituted by the Council in 1991, where longliners are prohibited from fishing within 50 nm of the NWHI and within 100 nm closed corridors connecting the non-contiguous closed circles. Further longline exclusion zones prohibit longline fishing in specific areas around the MHI (depending on the time of year and location, the exclusion zones around the MHI range from 25-75 nm). Because adult hawksbills are most likely foraging primarily in nearshore waters, the likelihood of an interaction with a longliner is very low. In addition, there have been no reported interactions between hawksbill turtles and other fisheries under the Pelagics FMP. Therefore, NMFS has determined that the proposed action is not likely to adversely affect hawksbill turtles.

Based on observed and reported interactions between the Hawaii-based longline fishery and four species of sea turtles, NMFS has determined that the proposed action is likely to adversely affect green, leatherback, loggerhead, and olive ridley turtles. Therefore, formal consultation is required in order to analyze the effects of the proposed action on these listed species.

The following subsections are synopses of the current state of knowledge on the life history, distribution, and population trends of these sea turtle species and that NMFS expects may be incidentally taken as a result of the proposed action. In addition, the Status of the Species and the Environmental Baseline, typically two separate sections in a Biological Opinion, are combined here because the status of the species and the factors affecting them are similar both within the action area and throughout their range in the Pacific Ocean.

### **Status of Listed Sea Turtles**

All stocks/populations of sea turtles adversely affected by the Pelagics FMP fisheries are in decline, except for olive ridleys and Hawaiian green turtles, which appear to be increasing. Impacts to sea

turtles in the Pacific Ocean are primarily due to the composite effect of human activities which include: the legal harvest and illegal poaching of adults, immatures, and eggs; incidental capture in fisheries (coastal and high-seas); and loss and degradation of nesting and foraging habitat as a result of coastal development, including predation by domestic dogs and pigs foraging on nesting beaches (associated with human settlement). Increased environmental contaminants (e.g. sewage, industrial discharge) and marine debris, which adversely impact nearshore ecosystems that turtles depend on for food and shelter, including sea grass and coral reef communities, also contribute to the overall decline. While it is generally accepted by turtle biologists and others that these factors are the primary cause of turtle population declines, in many cases there is a paucity of quantitative data on the magnitude of human-caused mortality. These four species of sea turtles are highly migratory or have a highly migratory phase in their life history, which makes them susceptible to being incidentally caught by fisheries operating throughout the Pacific Ocean. The Hawaii-based longline fishery under the Pelagics FMP is known to interact with all four species. In addition to anthropogenic factors, natural threats to the nesting beaches and pelagic-phase turtles such as coastal erosion, seasonal storms, predators, temperature variations, and phenomena such as El Niño also affect the survival and recovery of sea turtle populations. More information on the status of these species along with an assessment of overall impacts are found in this section as well as the Pacific Sea Turtle Recovery Plans (NMFS and USFWS, 1998a-d) and are reviewed extensively in Eckert (1993).

a. *Green Turtles*

Green turtles are thought to be declining throughout the Pacific Ocean, with the exception of Hawaii, as a direct consequence of a historical combination of overexploitation and habitat loss (Eckert, 1993). The species is listed as threatened under the ESA, except for breeding populations found in Florida and the Pacific coast of Mexico, which are listed as endangered. The International Union for Conservation of Nature and Natural Resources (IUCN) has classified the green turtle as “endangered”<sup>1</sup> due to an “observed, estimated, inferred or suspected reduction of at least 50% over the last 10 years or three generations, whichever is longer,” based on: (a) direct observation; (b) an index of abundance appropriate for the species; and (c) actual or potential levels of exploitation.

The genus *Chelonia* is composed of two taxonomic units at the population level, the eastern Pacific green turtle (referred to by some as “black turtle,” *C. mydas agassizii*), which ranges (including nesting) from Baja California south to Peru and west to the Galapagos Islands, and the nominate *C. m. mydas* in the rest of the range (insular tropical Pacific, including Hawaii).

Green turtles are distinguished from other sea turtles by their smooth carapace with four pairs of lateral scutes, a single pair of prefrontal scutes, and a lower jaw-edge that is coarsely serrated. Adult green turtles have a light to dark brown carapace, sometimes shaded with olive, and can exceed one meter in carapace length and 100 kilograms (kg) in body mass. Females nesting in Hawaii averaged 92 cm in

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<sup>1</sup>Under the IUCN, taxa are classified as endangered when they are not “critically endangered, but are facing a very high risk of extinction in the wild in the near future.

straight carapace length (SCL), while at the Olimarao Atoll in Yap, females averaged 104 cm in curved carapace length (CCL) and approximately 140 kg. In the rookeries of Michoacán, Mexico, females averaged 82 cm in CCL, while males averaged 77 cm CCL (*in* NMFS and USFWS, 1998a).

Green turtles are a circumglobal and highly migratory species, nesting mainly in tropical and subtropical regions. Based on growth rates observed in wild green turtles, skeletochronological studies, and capture-recapture studies, all in Hawaii, it is estimated that green turtles attain sexual maturity at an average age of at least 25 years (*in* Eckert, 1993). Growth rates and age to first reproduction in other north Pacific populations remain unquantified (Eckert, 1993). In Hawaii, green turtles lay up to six clutches of eggs per year (mean of 3.7), and clutches consist of about 100 eggs each. Females migrate to breed only once every two or possibly many more years. Eastern Pacific green turtles have reported nesting between two and six times during a season, laying a mean of between 65 and 86 eggs per clutch, depending on the area studied (Michoacan, Mexico and Playa Naranjo, Costa Rica (*in* Eckert, 1993 and NMFS and USFWS, 1998a).

The nonbreeding range of green turtles is generally tropical, and can extend approximately 500-800 miles from shore in certain regions (Eckert, 1993). They appear to prefer waters that usually remain around 20EC in the coldest month; for example, during warm spells (e.g., El Niño), green turtles may be found considerably north of their normal distribution. Stinson (1984) found green turtles to appear most frequently in U.S. coastal waters with temperatures exceeding 18EC. An east Pacific green turtle equipped with a satellite transmitter was tracked along the California coast and showed a distinct preference for waters with temperatures above 20EC (Eckert, unpublished data). Hawaiian green turtles monitored through satellite transmitters were found to travel more than 1,100 km from their nesting beach in the French Frigate Shoals, south and southwest against prevailing currents to numerous distant foraging grounds within the 2,400 kilometer span of the archipelago (Balazs, 1994; Balazs, *et al.*, 1994; Balazs and Ellis, 1996). Three green turtles outfitted with satellite tags on the Rose Atoll (the easternmost island at the Samoan Archipelago) traveled on a southwesterly course to Fiji, approximately 1,500 km distance (Balazs, *et al.*, 1994).

Tag returns of eastern Pacific green turtles establish that these turtles travel long distances between foraging and nesting grounds. In fact, 75 percent of tag recoveries from 1982-90 were from turtles that had traveled more than 1,000 kilometers from Michoacán, Mexico. Even though these turtles were found in coastal waters, the species is not confined to these areas, as indicated by 1990 sightings records from a NOAA research ship. Observers documented green turtles 1,000-2,000 statute miles from shore (Eckert, 1993). The east Pacific green is also the second-most sighted turtle in the east Pacific during tuna fishing cruises; they are frequent along a north-south band from 15EN to 5ES along 90EW, and between the Galapagos Islands and Central American Coast (NMFS and USFWS, 1998a). In a review of sea turtle sighting records from northern Baja California to Alaska, Stinson (1984) determined that the green turtle was the most commonly observed sea turtle on the U.S. Pacific Coast, with 62% reported in a band from southern California and southward. The northernmost reported resident population of green turtles occurs in San Diego Bay, where about 50-60 mature and immature turtles concentrate in the warm water effluent discharged by a power plant (McDonald, *et al.*, 1994). These turtles appear to have originated from east Pacific nesting beaches, based on

morphology and preliminary genetic analysis (*in* NMFS and USFWS, 1998a); however, the possibility exists that some are from Hawaii (P. Dutton, NMFS, personal communication, January, 2001). Stranding reports from the Hawaiian Islands from 1982-1999 indicate that the green turtle is the most commonly stranded sea turtle (96.5 percent, compared to other species), averaging around 150 per year (2,689 total/18 years).

Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, it is presumed that those in pelagic habitats live and feed at or near the ocean surface, and that their dives do not normally exceed several meters in depth (NMFS and USFWS, 1998a). The maximum recorded dive depth for an adult green turtle was 110 meters (Berkson, 1967, *in* Lutcavage and Lutz, 1997), while subadults routinely dive 20 meters for 9-23 minutes, with a maximum recorded dive of 66 minutes (Brill, *et al.*, 1995, *in* Lutcavage and Lutz, 1997). Additionally, it is presumed that drift lines or surface current convergences are preferential zones due to increased densities of likely food items. In the western Atlantic, drift lines commonly contain floating *Sargassum* capable of providing small turtles with shelter and sufficient buoyancy to raft upon (NMFS and USFWS, 1998a). Underwater resting sites include coral recesses, the underside of ledges, and sand bottom areas that are relatively free of strong currents and disturbance from natural predators and humans. In the MHI these foraging and resting areas for adults usually occur at depths greater than 10 meters, but probably not normally exceeding 40 meters. Available information indicates that green turtle resting areas are in proximity to their feeding pastures (NMFS, 2000e). Immature Hawaiian green turtles have been found in increasing numbers residing in “foraging pastures” around the eight main Hawaiian Islands. These pastures consist of a narrow band of shallow water around these islands and “accounts for 96% of the benthic habitat potentially available for recruitment by post-pelagic green turtles” (Balazs, 1996).

Although most green turtles appear to have a nearly exclusive herbivorous diet, consisting primarily of sea grass and algae (Wetherall *et al.*, 1993), those along the East Pacific coast seem to have a more carnivorous diet. Analysis of stomach contents of green turtles found off Peru revealed a large percentage of molluscs and polychaetes, while fish and fish eggs, and jellyfish and commensal amphipods comprised a lesser percentage (Bjorndal, 1997). In the Hawaiian Islands, green turtles are site-specific and consistently feed in the same areas on preferred substrates, which vary by location and between islands (*in* Landsberg, *et al.*, 1999).

In the western Pacific, the only major (> 2,000 nesting females) populations of green turtles occur in Australia and Malaysia. Pulau Redang, a coral fringed island located approximately 45 kilometers off the coast of Terengganu, Malaysia contains one of the largest green turtle rookeries in peninsular Malaysia, and a 1 nautical mile no-fishing zone has been established around the island to prevent interactions between fishing gear and internesting females (Liew and Chan, 1994). Smaller colonies occur in the insular Pacific islands of Polynesia, Micronesia, and Malaysia (Wetherall *et al.*, 1993). In Taiwan, Cheng and Chen (1996) report that between 1992 and 1994, green turtles were found nesting on 9 of 11 beaches on Wan-Am Island (Peng-Hu Archipelago). The numbers, however, were small, between 8 and 14 females nested during each of these 3 years.

In Japan, the Ogasawara Islands, located approximately 1,000 km south of Tokyo, serve as the

northern edge of green turtles rookeries. In the late 1800s, when Japan first colonized the islands, the government encouraged a sea turtle fishery. Declines in catch were steady from 1880-1890s (1,000-1,800 adults taken annually) through the mid-1920s (250 taken annually). Data from 1945-1972 (American occupation) indicate that 20-80 turtles were taken annually, and since then, annual harvests have fluctuated from 45-225 turtles per year (Horikoshi, *et al.*, 1994) (Suganuma, *et al.* (1996) estimates 100 mating adults are speared by fishermen annually). Beach census data from 1985-93 indicate that 170-649 clutches were deposited each year (43 to 162 nesting females, assuming a female deposited 4 clutches during a nesting season). The Ogasawara population has declined in part due to past commercial exploitation, and it is likely to continue if fishery effort continues (Horikoshi, *et al.*, 1994).

Thousands of islands comprise the eight U.S.-affiliated Pacific island groups, and of the sea turtle species, greens and hawksbills make up most of the composition. Unfortunately, there is a serious shortage of information on the population sizes, distribution, and migration patterns of these turtles, which can hamper recovery efforts. Based on limited data, green turtle populations in the Pacific islands have declined dramatically, due foremost to harvest of eggs and adults by humans. In the green turtle recovery plans, directed take of eggs and turtles was identified as a “major problem” in the American Samoa, Guam, Palau, Commonwealth of the Northern Mariana Islands, Federated States of Micronesia, Republic of the Marshall Islands, and the Unincorporated Islands (Wake, Johnston, Kingman, Palmyra, Jarvis, Howland, Baker, and Midway). Severe overharvests have resulted in modern times from a number of factors: 1) the loss of traditional restrictions limiting the number of turtles taken by island residents; 2) modernized hunting gear; 3) easier boat access to remote islands; 4) extensive commercial exploitation for turtle products in both domestic markets and international trade; 5) loss of the spiritual significance of turtles; 6) inadequate regulations; and 7) lack of enforcement (NMFS and USFWS, 1998a).

In Hawaii, green turtles nest on six small sand islands at French Frigate Shoals, a long atoll situated in the middle of the Hawaiian Archipelago (Balazs, 1995). Unlike any other regional sea turtle populations, green turtles in Hawaii are genetically distinct and geographically isolated. Ninety percent of the nesting and breeding activity of the Hawaiian green turtle occurs at the French Frigate Shoals, where 200-700 females are estimated to nest annually (NMFS and USFWS, 1998a). Important resident areas have been identified and are being monitored along the coastlines of Oahu, Molokai, Maui, Lanai, Hawaii, and at large nesting areas in the reefs surrounding the French Frigate Shoals, Lisianski Island, and Pearl and Hermes Reef (Balazs, 1982; Balazs *et al.*, 1987). Since the establishment of the ESA in 1973, and following years of exploitation, the nesting population of Hawaiian green turtles has shown a gradual but definite increase (Balazs, 1996). For example, the number of green turtles nesting at an index study site at East Island has tripled since systematic monitoring began in 1973 (NMFS and USFWS, 1998a). However, the green turtle population in this area is afflicted with a tumor disease, fibropapilloma, which is of an unknown etiology and often fatal, as well as spirochidiasis, both of which are the major causes of strandings of this species (G. Balazs, NMFS, personal communication, 2000). The presence of fibropapillomatosis among stranded turtles has increased significantly over the past 17 years, ranging from 47-69 percent during the past decade (Murakawa, *et al.*, 2000). Green turtles captured off Molokai from 1982-96 showed a massive

increase in the disease over this period, peaking at 61% prevalence in 1995 (Balazs, *et al.*, 1998). Preliminary evidence suggests that there is an association between the distribution of fibropapillomatosis in the Hawaiian Islands and the distribution of toxic benthic dinoflagellates (*Prorocentrum* spp.) known to produce a tumor promoter, okadaic acid (Landsberg, *et al.*, 1999). Fibropapillomatosis is considered an inhibiting factor to the full recovery of the Hawaiian green turtle populations, and the incidence of decreased growth rates in afflicted turtles is a minimum estimate of the impact of the disease (Balazs, *et al.*, 1998).

The primary green turtle nesting grounds in the eastern Pacific are located in Michoacán, Mexico, and the Galapagos Islands, Ecuador (NMFS and USFWS, 1998a). Here, green turtles were widespread and abundant prior to commercial exploitation and uncontrolled subsistence harvest of nesters and eggs. More than 165,000 turtles were harvested from 1965 to 1977 in the Mexican Pacific. In the early 1970s nearly 100,000 eggs per night were collected from these nesting beaches (*in* NMFS and USFWS, 1998a). The nesting population at the two main nesting beaches in Michoacán (Colola, responsible for 70% of total green turtle nesting in Michoacán (Delgado and Alverado, 1999) and Maruata) decreased from 5,585 females in 1982 to 940 in 1984. Despite long-term protection of females and their eggs at these sites since 1990, the population continues to decline, and it is believed that adverse impacts (including incidental take in various coastal fisheries as well as illegal directed take at forage areas) continue to prevent recovery of endangered populations (P. Dutton, NMFS, personal communication, 1999; W. Nichols, University of Arizona, personal communication, 2000). In addition, the black market for sea turtle eggs in Mexico has remained as brisk as before the ban (Delgado and Alvarado, 1999). On Colola, an estimated 500-1,000 females nested nightly in the late 1960s. In the 1990s, that number dropped to 60-100 per night, or about 800-1,000 turtles per year (Eckert, 1993). During the 1998-99 season, based on a comparison of nest counts and egg collection data, an estimated 600 greens nested at Colola. Although only about 5% of the nests were poached at Colola during this season, approximately 50% of the nests at Maruata were poached, primarily because of difficulties in providing protections as a result of political infighting (Delgado and Alvarado, 1999).

There are few historical records of abundance of green turtles from the Galapagos - only residents are allowed to harvest turtles for subsistence, and egg poaching occurs only occasionally. An annual average of 1,400 nesting females was estimated for the period 1976-1982 in the Galapagos Islands (NMFS and USFWS, 1998a).

Green turtles encountered by U.S. vessels fishing managed under the Pelagics FMP may originate from a number of known proximal, or even distant, breeding colonies in the region. Genetic sampling of green turtles taken by the Hawaii-based longline fishery indicates representation from nesting beaches on Hawaii (French Frigate Shoals) and the Pacific coast of Mexico population. Preliminary genetic analysis revealed that of eight greens caught by the Hawaii-based longline fishery, four were of eastern Pacific (Mexico) origin, three were of eastern Pacific or Hawaiian origin, and one was of Hawaiian origin (P. Dutton, NMFS, personal communication, January, 2001).

b. *Leatherback Turtles*



The leatherback turtle is listed as endangered under the ESA throughout its global range. Furthermore, the Red List 2000 of the IUCN has classified the leatherback as “critically endangered”<sup>2</sup> due to “an observed, estimated, inferred or suspected reduction of at least 80% over the last 10 years or three generations, whichever is the longer,” based on: (a) direct observation; (b) an index of abundance appropriate for the taxon; and (c) actual or potential levels of exploitation. Increases in the number of nesting females have been noted at some sites *in the Atlantic*, but these are far outweighed by local extinctions, especially of island populations, and the demise of once large populations *throughout the Pacific*, such as in Malaysia and Mexico. Spotila *et al.* (1996) estimated the global population of female leatherback turtles to be only 34,500 (confidence limits: 26,200 to 42,900) nesting females; however, the eastern Pacific population has continued to decline since that estimate, leading some researchers to conclude that the leatherback is now on the verge of extinction in the Pacific Ocean (e.g. Spotila, *et al.*, 1996; Spotila, *et al.*, 2000).

Leatherback turtles are the largest of the marine turtles, with a CCL often exceeding 150 cm and front flippers that are proportionately larger than in other sea turtles and may span 270 cm in an adult (NMFS and USFWS, 1998b). In view of its unusual ecology, the leatherback is morphologically and physiologically distinct from other sea turtles. Its streamlined body, with a smooth, dermis-sheathed carapace and dorso-longitudinal ridges may improve laminar flow of this highly pelagic species. Adult females nesting in Michoacán, Mexico averaged 145 cm CCL (L. Sarti, Universidad Nacional Autonoma de Mexico, unpublished data, *in* NMFS and USFWS, 1998b), while adult female leatherback turtles nesting in eastern Australia averaged 162 cm CCL (Limpus, *et al.*, 1984, *in* NMFS and USFWS, 1998b).

Leatherback turtles have the most extensive range of any living reptile and have been reported circumglobally from 71°N to 42°S latitude in the pelagic Pacific and in all other major pelagic ocean habitats (NMFS and USFWS, 1998b). For this reason, however, studies of their abundance, life history and ecology, and pelagic distribution are exceedingly difficult. Similar to the olive ridley turtle, leatherback turtles lead a completely pelagic existence, foraging widely in temperate waters except during the nesting season, when gravid females return to tropical beaches to lay eggs. Males are only rarely observed near nesting areas, and it has been proposed that mating most likely takes place outside of the tropical waters, before females move to their nesting beaches (Eckert and Eckert, 1988). They are highly migratory, exploiting convergence zones and upwelling areas in the open ocean, along continental margins, and in archipelagic waters (Morreale, *et al.*, 1994; Eckert, 1998; Eckert, 1999a). In a single year, a leatherback may swim more than 10,000 kilometers (Eckert, 1998).

Recent satellite telemetry studies indicate that adult leatherback turtles follow bathymetric contours over their long pelagic migrations and typically feed on cnidarians (jellyfish and siphonophores) and tunicates (pyrosomas and salps), and their commensals, parasites and prey (NMFS and USFWS, 1998b). Because of the low nutritive value of jellyfish and tunicates, it has been estimated that an adult

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<sup>2</sup>Taxa are categorized as critically endangered when they are facing an extremely high risk of extinction in the wild in the immediate future.

leatherback would need to eat about 50 large jellyfish (equivalent to approximately 200 liters) per day to maintain its nutritional needs (Duron, 1978, *in* Bjorndal, 1997). Compared to greens and loggerheads, who consume approximately 3-5% of their body weight per day, leatherback turtles may consume perhaps 20-30% of their body weight per day (Davenport and Balazs, 1991). Surface feeding has been reported in U.S. waters, especially off the west coast (Eisenberg and Frazier, 1983), but foraging may also occur at depth. Based on offshore studies of diving by adult females nesting on St. Croix, U.S. Virgin Islands, Eckert *et al.* (1989) proposed that observed internesting<sup>3</sup> dive behavior reflected nocturnal feeding within the deep scattering layer (strata comprised primarily of vertically migrating zooplankton, chiefly siphonophore and salp colonies, as well as medusae). Hartog (1980, *in* NMFS and USFWS, 1998b) also speculated that foraging may occur at depth, when nematocysts from deep water siphonophores were found in leatherback stomach samples. Davenport (1988, *in* Davenport and Balazs, 1991) speculated that leatherback turtles may locate pyrosomas at night due to their bioluminescence; however direct evidence is lacking.

Leatherback turtles also appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting that maximum exploitation of the water column is of paramount importance to the leatherback (Eckert, *et al.*, 1989). Maximum dive depths for post-nesting females in the Caribbean have been recorded at 475 meters and over 1,000 meters, with routine dives recorded at between 50 and 84 meters. The maximum dive length recorded for such female leatherback turtles was 37.4 minutes, while routine dives ranged from 4-14.5 minutes (*in* Lutcavage and Lutz, 1997). A total of six adult female leatherback turtles from Playa Grande, Costa Rica were monitored at sea during their internesting intervals and during the 1995 through 1998 nesting seasons. The turtles dived continuously for the majority of their time at sea, spending 57-68% of their time submerged. Mean dive depth was  $19 \pm 1$  meters and the mean dive duration was  $7.4 \pm 0.6$  minutes (Southwood, *et al.*, 1999). Migrating leatherback turtles also spend a majority of time at sea submerged, and they display a pattern of continual diving (Standora, *et al.*, 1984, *in* Southwood, *et al.*, 1999). Eckert (1999a) placed transmitters on nine leatherback females nesting at Mexiquillo Beach and recorded dive behavior during the nesting season. The majority of the dives were less than 150 meters depth, although maximum depths ranged from 132 meters to over 750 meters. Although the dive durations varied between individuals, the majority of them made a large proportion of very short dives (less than two minutes), although Eckert speculates that the short duration dives most likely represent surfacing activity after each dives. Excluding these short dives, five of the turtles preferred dive durations greater than 24 minutes, while three others preferred dive durations between 12-16 minutes.

On the Pacific coast of Mexico, female leatherback turtles lay 1-11 clutches per year (mean=5.7), with clutch size averaging 64 yolked eggs (each clutch contains a complement of yolckless eggs, sometimes comprising as much as 50 percent of total clutch size, a unique phenomenon among leatherback turtles and some hawksbills (Hirth and Ogren, 1987)). Clutch sizes in Terengganu, Malaysia, and in Pacific Australia were larger, averaging around 85-95 yolcked eggs and 83 yolcked eggs, respectively (*in* Eckert, 1993). Females are believed to migrate long distances between foraging and breeding grounds,

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<sup>3</sup>Internesting – time spent between laying clutches of eggs during a single nesting season.

at intervals of typically two or three years (García and Sarti, 2000). Spotila *et al.* (2000), found the mean re-nesting interval of females on Playa Grande, Costa Rica to be 3.7 years, while in Mexico, 3 years was the typical reported interval (L. Sarti, personal communication, 2000). In Mexico, the nesting season generally extends from November to February, although some females arrive as early as August (Sarti *et al.*, 1989). In the western Pacific, nesting peaks on Jamursba-Medi Beach (Irian Jaya) from May to August, on War Mon Beach (also Irian Jaya) from November to January (Starbird and Suarez, 1994), in peninsular Malaysia in June and July (Chan and Liew, 1989), and in Queensland, Australia in December and January (Limpus and Riemer, 1984).

Using skeletochronological analysis of a small sample size of leatherback sclerotic ossicles, Zug and Parham (1996) suggested that mean age at sexual maturity for leatherback turtles is around 13 to 14 years, giving them the highest juvenile growth rate of all sea turtle species. Zug and Parham (1996) concluded that for conservation and management purposes, 9 years is a likely minimum age for maturity of leatherback turtles, based on the youngest adult in their sample. The natural longevity of leatherback turtles has not been determined (NMFS and USFWS, 1998b), although there are recorded documentations of post-maturation survival on the order of about 20 years (Pritchard, 1996).

Migratory routes of leatherback turtles originating from eastern and western Pacific nesting beaches are not entirely known. However, satellite tracking of post-nesting females and genetic analyses of leatherback turtles caught in U.S. Pacific fisheries or stranded on the west coast of the U.S. present some strong insight into at least a portion of their routes and the importance of particular foraging areas. Current data from genetic research suggest that Pacific leatherback stock structure (natal origins) may vary by region. Because leatherback turtles are highly migratory and stocks mix in high seas foraging areas, and based on genetic analyses of samples collected by Hawaii-based longline observers, leatherback turtles inhabiting the action area are comprised of individuals originating from nesting assemblages located south of the equator in Indonesia and in the eastern Pacific along the Americas (e.g., Mexico, Costa Rica) (Dutton, *et al.*, 2000).

For female leatherback turtles nesting at Mexiquillo Beach, Mexico, the eastern Pacific region has been shown to be a critical migratory route. Nine females outfitted with satellite transmitters in 1997 traveled along almost identical pathways away from the nesting beach. These individuals moved south and, upon encountering the North Equatorial Current at about 8°N, diverted west for approximately 800 km and then moved east/southeast towards the waters off Peru and Chile (Eckert, 1999a). In addition, four leatherback turtles recovered from Chilean fishing vessels from 1988-91 had been tagged on nesting beaches in Costa Rica and Mexico (Brito-Montero, 1995, *in* Donoso, 2000).

Morreale *et al.* (1994) demonstrated that satellite tagged, post-nesting leatherback turtles leaving Costa Rica followed precisely defined, long-distance migratory pathways after nesting. Despite differences in dates of departure from the nesting areas, nesting cohorts followed along nearly identical pathways. All 6 leatherback turtles' (from the Pacific and Caribbean coasts of Costa Rica) movements paralleled deepwater bathymetric contours ranging from 200-3,500 meters. When a turtle's path intersected an abyssal plain, it veered along the outer slope, and when an abyssal plain was unavoidable, the turtle crossed it at its narrowest point. These studies underscore the importance of this

offshore habitat and migratory corridors and the likelihood that sea turtles are present on fishing grounds, particularly for large commercial fishing fleets south of the equator (Eckert, 1997). Eckert (1999a) speculates that leatherback turtles leaving the nesting areas of Mexico and Costa Rica may be resource-stressed by a long reproductive season with limited food and the high energetic requirements brought about by the demands of reproduction, elevated water temperatures, or both. When they leave, their greatest need is to replenish energy stores (e.g. fat) and they must move to areas where food is concentrated (e.g. upwelling areas). These eastern Pacific nesting stocks may also move northwest, as genetic samples from two leatherback turtles caught south of the main Hawaiian islands by the Hawaii-based longline fishery indicated representation from eastern Pacific nesting beaches (Dutton *et al.*, in press, and unpublished).

Migratory corridors of leatherback turtles originating from western Pacific nesting beaches most likely exist along the eastern seabords of Australia, Asia and the former Soviet Union (NMFS and USFWS, 1998b). Genetic markers in 12 of 14 leatherback turtles sampled to date from the central North Pacific (captured in the Hawaii-based longline fishery) have identified those turtles as originating from nesting populations in the southwestern Pacific; the other 2 specimens, taken in the southern range of the Hawaii fishery, were from nesting beaches in the eastern Pacific (P. Dutton, NMFS, personal communication, January, 2001). Stranding records from 1982-99 indicate that the leatherback rarely strands in the Hawaiian Islands; only five leatherback turtles have been recorded stranded in 18 years (G. Balazs, NMFS, personal communication, 2000).

Leatherback turtles originating from western Pacific beaches have also been found along the U.S. mainland. Here, leatherback turtles have been sighted and reported stranded as far north as Alaska (60EN) and as far south as San Diego, California (NMFS and USFWS, 1998b). Of the stranded leatherback turtles that have been sampled to date from the U.S. mainland, all have been of western Pacific nesting stock origin (P. Dutton, NMFS, personal communication, 2000). Genetic analysis of samples from two leatherback turtles taken off California and Oregon by the CA/OR drift gillnet fishery revealed that they both originated from western Pacific nesting beaches (i.e. Indonesia/Solomon Islands/Malaysia) (P. Dutton, NMFS, personal communication, March, 2000). In addition, two female leatherback turtles were recently captured and tagged in Monterey Bay, California, on September 7-8, 2000 and fitted with transmitters. One of these individuals was of a size normally associated with the western Pacific nesting stock, which are, on average, 10-20 centimeters larger than eastern Pacific nesting stocks (Zug and Parham, 1996). As of 9/21/00, both were on a southwest migratory path and appeared to be heading to the western Pacific nesting beaches. On 11/17/00, the larger female stopped transmitting when it entered an area southeast of the Hawaiian Islands (145EW longitude, 15EN latitude). The other leatherback continues to travel west along the North Equatorial Current towards Indonesia, and as of 1/18/01, it was located around 167EW longitude, 7EN latitude (P. Dutton, NMFS, and S. Eckert, HSWRI, personal communication, January, 2001).

Lastly, genetic analyses of two leatherback turtles taken by fishing vessels in Chilean waters suggest that one is from a western Pacific or Indian Pacific nesting population and the other is of eastern Pacific origin. This is the first evidence that leatherback turtles from western Pacific nesting beaches occur in Chilean waters, confirming transoceanic migration to eastern Pacific forage areas in the southern

hemisphere (Donoso, *et al.*, 2000).

Hawaiian fishermen in offshore waters have seen leatherback turtles generally beyond 100 fathoms, but within sight of land. Two areas where sightings have taken place are off the north coast of Oahu and the west coast of the Island of Hawaii. The pelagic zone surrounding the Hawaiian Islands apparently is regularly used as foraging habitat and migratory pathways for this species (NMFS, 1991).

The distribution of juvenile leatherback turtles has long been a mystery. However, a recent compilation and analysis of sighting and stranding data for the species has yielded some interesting insight into the developmental habitats of this species at earlier life stages. It appears that young leatherback turtles (carapace length <100cm) reside only in waters warmer than 26EC, which should generally place them outside of areas in which longline swordfish fleets operate (Eckert, 1999b; Eckert, submitted manuscript). However, as discussed further in the Effects of the Action section, the Hawaii-based longline fishery has been observed to take a few subadult leatherback turtles (straight carapace length < 100 cm).

Based on published estimates of nesting female abundance, leatherback populations are declining at all major Pacific basin nesting beaches, particularly in the last two decades (Spotila *et al.*, 1996; NMFS and USFWS, 1998b; Spotila, *et al.*, 2000). Declines in nesting populations have been documented through systematic beach counts or surveys in Malaysia (Rantau Abang, Terengganu), Mexico and Costa Rica. In other leatherback nesting areas, such as Irian Jaya and the Solomon Islands, there have been no systematic consistent nesting surveys, so it is difficult to assess the status and trends of leatherback turtles at these beaches. In all areas where leatherback nesting has been documented, however, current nesting populations are reported by scientists, government officials, and local observers to be well below abundance levels of several decades ago. The collapse of these nesting populations was most likely precipitated by a tremendous overharvest of eggs coupled with incidental mortality from fishing (Sarti *et al.*, 1996; Eckert, 1997).

#### *Eastern Pacific nesting populations of leatherback turtles*

Leatherback nesting populations are declining at a rapid rate along the Pacific coast of Mexico and Costa Rica (see Appendix C, Table 1). At Las Baulas National Park, Costa Rica, the number of nesting leatherback turtles has declined from 1,500 in 1988-1989 to 193 in 1993-1994 (Steyermark *et al.*, 1996). Leatherback turtles have been studied at Playa Grande (in Las Baulas), the fourth largest leatherback nesting colony in the world, since 1988. During the 1988-89 season (July-June), 1,367 leatherback turtles nested on this beach, and by the 1998-99 season, only 117 leatherback turtles nested (Spotila, 2000). The 1999-2000 season showed a slight increase in the number of adult females nesting here, with slightly over 200 nesting (preliminary data presented on the Las Baulas leatherback conservation project website<sup>4</sup>). During the last three nesting seasons (1996 through 1999), an average of only 25% of the turtles were remigrants (turtles returning to nest that were observed nesting in previous nesting seasons). Less than 20% of the turtles tagged in 1993 through 1995 returned to nest

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<sup>4</sup>[http://www.leatherback.org/lasbaulas/costa-rica/Las\\_Baulas/Results\\_progress/1999\\_00/Index.html](http://www.leatherback.org/lasbaulas/costa-rica/Las_Baulas/Results_progress/1999_00/Index.html)

in the next five years (Spotila, *et al.*, 2000). Remigration intervals for leatherback turtles at nesting beaches in South Africa and the U.S. Caribbean have been documented as over 91% returning within 5 years or less (Hughes, 1996 and Boulon, *et al.* 1996 in Spotila, *et al.*, 2000). Comparatively few leatherback turtles are returning to nest on east Pacific nesting beaches and it is likely that leatherback turtles are experiencing abnormally high mortalities during non-nesting years. Since 1993, environmental education and conservation efforts through active law enforcement has greatly reduced egg poaching in Costa Rica (Chaves, *et al.*, 1996). For example, during the 1993-94 nesting season, poaching accounted for only 1.3 percent of the loss of nests on Playa Grande. Other losses were due to predation, tidal effects and failure in egg development or infestation by maggots (Schwandt, *et al.*, 1996).

The decline of leatherback subpopulations is even more dramatic off Mexico. According to reports from the late 1970s and early 1980s, three beaches located on the Pacific coast of Mexico sustained a large portion of all global nesting of leatherback turtles, perhaps as much as one-half. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherback turtles has declined from 70,000<sup>5</sup> in 1982 (Pritchard, 1982, in Spotila *et al.*, 1996) to slightly more than 200 adult females during the last two seasons (1997-98 and 1998-1999 (Sarti *et al.*, 2000). Monitoring of the nesting assemblage at Mexiquillo, Mexico has been continuous since 1983-84. According to Sarti *et al.* (1996), nesting declined at this location at an annual rate of over 22 percent from 1984 to 1995. Sarti *et al.* (1998) reports:

“While reporting the results for the 1995-96 nesting season (Sarti *et al.*, 1996), we regarded beaches having densities higher than 50 nests per kilometer as the most important. In the present season [1997-98] no beach reached such density values: the main beaches had 5 or more nests per kilometer, and none were higher than 25. This is evidence of the large decrement witnessed from the start of the aerial surveys, and may indicate that the nesting population still has a declining trend despite the protection efforts in the major beaches.”

Furthermore, Sarti, *et al.* (2000) notes that during the 1980s, 30% of the nesting females per season were remigrants, but since the mid-1990s, there has been no evidence of remigration, even with more efficient tagging methods.

Although the causes of the decline in the nesting populations are not entirely clear, Sarti *et al.* (1998) surmises that the decline could be a result of intensive egg poaching in the nesting areas, incidental capture of adults or juveniles in high seas fisheries, and natural fluctuations due to changing environmental conditions. Although leatherback turtles are not generally captured for their meat or skin

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<sup>5</sup>This estimate of 70,000 adult female leatherback turtles comes from a brief aerial survey of beaches by Pritchard (1982), who has commented: “I probably chanced to hit an unusually good nesting year during my 1980 flight along the Mexican Pacific coast, the population estimates derived from which (Pritchard, 1982) have possibly been used as baseline data for subsequent estimates to a greater degree than the quality of the data would justify” (Pritchard, 1996).

in Mexico, the slaughter of female leatherback turtles has been detected on beaches such as Piedra de Tiacoyunque, Guerrero (Sarti, *et al.*, 2000). In addition, there is little information on incidental capture of adults due to coastal fisheries off Mexico, but entanglement in longlines and driftnets probably account for some mortality of leatherback turtles. Eckert (1997) speculates that the swordfish gillnet fisheries in Peru and Chile have contributed to the decline of the leatherback in the eastern Pacific. The decline in the nesting population at Mexiquillo, Mexico occurred at the same time that effort doubled in the Chilean driftnet fishery.

Most conservation programs aimed at protecting nesting sea turtles in Mexico have continued since the early 1980s, and there is little information on the degree of poaching prior to the establishment of these programs. However, Sarti *et al.* (1998) estimates that as much as 100% of the clutches were taken from the Mexican beaches. Since protective measures have been in place, particularly emergency measures recommended by a joint U.S./Mexico leatherback working group meeting in 1999, there has been greater nest protection and nest success (Table III-1). Mexican military personnel were present during the 1999-2000 season at three of the primary nesting beaches in Mexico (Llano Grande, Mexiquillo, and Tierra Colorado), responsible for approximately 34% of all nesting activity in Mexico. Of 1,294 nests documented, 736 were protected (57%), resulting in a total of 25,802 hatchlings. Monitoring and protection measures at two secondary nesting beaches resulted in the protection of 67% and 10% at Barra de la Cruz and Playa Ventura, respectively. Currently, the primary management objective is to protect over 95% of nests laid at the three index beaches (includes protecting nesting females, eliminating illegal egg harvest, and relocating nests to protected hatcheries) and to maximize protection of all the secondary nesting beaches over the next three years. NMFS has committed funding for the next three years to help implement these objectives (minutes from joint U.S./Mexico Leatherback Working Group meeting, 23-24 May, 2000).

**Table III-1. Nest protection at index beaches on the Pacific coast of Mexico (Source: Sarti *et al.*, personal communication, 2000)**

Season	Number of clutches laid	Number of clutches protected	Percentage of clutches protected
1996-97	445	86	19.3%
1997-98	508	101	19.9%
1998-99	442	150	33.9%
1999-00	1590	943	58.7%

From tagging and aerial surveys, Spotila *et al.* (2000) have estimated that there are currently 687 adult females and 518 subadults comprising the Central American population of leatherback turtles. With an estimated Mexican population of 1,000 adults and 750 subadults (by Spotila *et al.*, 2000), the entire east Pacific leatherback population has been estimated by Spotila *et al.* (2000) to contain approximately 2,955 females (1,687 adults and 1,268 subadults); however, insufficient foundation was given for these estimates (i.e. derivation of estimates are unclear, and models rely on theoretical

assumptions that need further evaluation and testing).

#### *Western Pacific Populations of Leatherback Turtles*

Similar to their eastern Pacific counterparts, leatherback turtles originating from the western Pacific are also threatened by poaching of eggs, killing of nesting females, human encroachment on nesting beaches, incidental capture in fishing gear, beach erosion, and egg predation by animals. Little is known about the status of the western Pacific leatherback nesting populations but once major leatherback nesting assemblages are declining along the coasts of Malaysia, Indonesia and the Solomon Islands. Low density and scattered nesting of leatherback turtles occurs in Fiji, Thailand, Australia (primarily western and to a lesser extent, eastern), and Papua New Guinea. In the Solomon Islands, the rookery size is estimated to be less than 100 females nesting per year (D. Broderick, personal communication, *in* Dutton, *et al.*, 1999). In Indonesia, low density nesting occurs along western Sumatra (200 females nesting annually) and in southeastern Java (50 females nesting annually), although the last known information is from the early 1980s (*in* Suarez and Starbird, 1996a). The largest extant leatherback rookery in the Indo-Pacific lies on the north Vogelkop coast of Irian Jaya, with over 1,000 females nesting during the 1996 season (Suarez *et al.*, *in press*) (see Table III-3).

As with the eastern Pacific nesting populations, the decline of leatherback turtles is severe at one of the most significant nesting sites in the region - Terengganu, Malaysia, with current nesting representing less than 2 percent of the levels recorded in the 1950s, and the decline is continuing. The nesting population at this location has declined from 3,103 females estimated nesting in 1968 to 2 nesting females in 1994 (Chan and Liew, 1996) (Table III-2). With one or two females reportedly nesting each year, this population has essentially been eradicated (P. Dutton, personal communication, 2000). Years of excessive egg harvest, egg poaching, the direct harvest of adults in this area, as well as incidental capture in various fisheries in territorial and international waters, have impacted the Malaysian population of leatherback turtles. There were two periods in which there were sharp declines in nesting leatherback turtles at this location: 1972-74 and 1978-80. Between 1972 and 1974, the number of females nesting declined 21% and coincided with a period of rapid development in the fishing industry, particularly trawling, in Terengganu (Chan *et al.*, 1988 *in* Chan and Liew, 1996). Between 1978 and 1980, nestings dropped an average of 31% annually, and coincided directly with the introduction of the Japanese high seas squid fishery of the North Pacific in 1978 (Yatsu *et al.*, 1991, *in* Chan and Liew, 1996). Because tagged individuals from Rantau Abang have been recovered from as far away as Taiwan, Japan and Hawaii, this fishery, as well as fisheries operating within the South China Sea, may have impacted the Malaysian leatherback population (Chan and Liew, 1996). After 1980, rates of decline averaged 16% annually, suggesting continuing threats from fisheries (Chan and Liew, 1996).

Table III-2. Number of nesting females per year in Terengganu, Malaysia (summarized in Spotilla, <i>et al.</i> , 1996)											
1968	1970	1972	1974	1976	1978	1980	1984	1987	1988	1993	1994
3,103	1,760	2,926	1,377	1,067	600	200	100	84	62	20	2

The nesting populations of leatherback turtles in Irian Jaya, Indonesia appear to be steady, although



without systematic consistent surveys of nesting beaches, an reliable assessment of the trends and status of leatherback turtles here is difficult. However, there has yet been no evidence of the collapse documented in Malaysia or the in the eastern Pacific. Leatherback nesting generally takes place on two major beaches, located 30 km apart, on the north Vogelkop coast of Irian Jaya, Jamursba-Medi (18 km) and War-Mon beach (4.5 km) (Starbird and Suarez, 1994). As shown in Table III-3, Suarez, *et al.* (in press) has compiled, re-analyzed, and standardized data collected from leatherback nesting surveys in the 1980s and 1990s. In addition, Suarez *et al.* (in press) has included information on the estimated number of nests lost due to both natural and anthropogenic causes. For example, during 1984 and 1985, on Jamursba-Medi, 40-60% of nests were lost to inundation and erosion, while 90% of those nests not taken by poachers<sup>6</sup> or by the sea were destroyed by feral pigs (*Sus scrofa*). Eggs from poached nests were commercially harvested for sale in the Sarong markets until 1993, when the beaches first received protection by the Indonesian government (J. Bakarbesy, personal communication, *in* Suarez and Starbird, 1996a). During the 1993-96 seasons, environmental education activities in nearby villages and protection measures on this same beach were put into place, with unreported results. Again, approximately 90% of those nests not taken by poachers or the sea<sup>7</sup> were destroyed by pigs (Suarez *et al.* in press). War-Mon beach supports a lower percentage of nesting females, yet egg poaching for subsistence accounted for over 60% of total nest loss during 1993-94, and total loss of nests due to pig predation was 40% (because there are more people in this region, there is more pig hunting; hence less pig predation of leatherback eggs (Starbird and Suarez, 1994)).

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<sup>6</sup>Suarez, *et al.* (in press) provided no information on the estimated percentage of nests lost to poachers.

<sup>7</sup>No information on percentage of nests lost to poachers of the sea or were given, except that it was “noted.”

**Table III-3. Estimated numbers of female leatherback turtles nesting along the north coast of Irian Jaya (Summarized by Suarez, *et al.*, in press.)**

Survey Period	# of Nests	Adjusted # Nests	Estimated # of Females <sup>3</sup>
Jamursba-Medi Beach:			
September, 1981	4,000+	7,173 <sup>1</sup>	1,232-1,623
April - Oct. 1984	13,360	13,360	2,303-3036
April - Oct. 1985	3,000	3,000	[(658)-731]
June - Sept. 1993	3,247	4,329 <sup>2</sup>	746-984
June - Sept. 1994	3,298	4,397 <sup>2</sup>	758-999
June - Sept. 1995	3,382	4,509 <sup>2</sup>	777-1025
June - Sept., 1996	5,058	6,744 <sup>2</sup>	1,163-1,533
War-Mon Beach:			
Nov. 1984 - Jan. 1985	1,012	N/A	175-230
Dec. 1993	406	653	128 - 169

<sup>1</sup>The total number of nests reported during aerial surveys were adjusted to account for loss of nests prior to the survey. Based on data from other surveys on Jamursba-Medi, on average 44% of all nests are lost by the end of August.

<sup>2</sup>The total number of nests have been adjusted based on data from Bhaskar's surveys from 1984-85 from which it was determined that 25% of the total number of nests laid during the season (4/1-10/1) are laid between April and May.

<sup>3</sup>Based on Bhaskar's tagging data, an average number of nests laid by leatherback turtles on Jamursba-Medi in 1985 was 4.4 nests per female. This is consistent with estimates for the average number of nests by leatherback turtles during a season on beaches in Pacific Mexico, which range from 4.4 to 5.8 nests per female (Sarti *et al.*, unpub. report). The range of the number of females is estimated using these data.

Recently, monitoring of leatherback turtles nesting at Jamursba Medi revealed that a total of 3,408 adult female nesting activities were recorded between May and October, 1999, and of these, 3,244 resulted in egg laying. Peak nesting occurred in July, 1999, when approximately 30 leatherback nests were recorded nightly (Putrawidjaja, 2000). Given this, without adjusting the total observed to include the month of April (see footnote 2 in Table III-3) and assuming that the average number of nests per female ranged between 4.4 to 5.8 (see footnote 3 in Table III-3), the number of females nesting during 1999 was between 586 and 775 leatherback turtles, approximately half of the number observed nesting during 1996.

In the Kai Islands (also spelled "Kei Islands"), located approximately 1,000 kilometers southwest of the Irian Jaya nesting beaches, adult leatherback turtles are traditionally hunted and captured at sea by local people. Villagers hunt leatherback turtles only for ritual and subsistence purposes, and, according to their beliefs (known as *adat*), they are forbidden to sell or trade the meat. However, due to population increase and deforestation of the area which has lead to the loss of forest resources such as

deer, pigs, and birds, villagers are taking leatherback turtles more for their increased need for meat for subsistence than for traditional purposes (Suarez and Starbird, 1996b). The carapace is rendered for oil, and the meat from the plastron is shared among villagers (Starbird and Suarez, 1994). Based on a study conducted during October-November, 1994, Suarez and Starbird (1996a) estimated that approximately 87 leatherback turtles were taken annually by villagers in the Kai Islands, and this estimate did not include incidental take by local gill and shark nets. Locals report that sea turtle populations in the area have declined dramatically (Suarez, 1999). Overall, approximately 200 leatherback turtles, both adult males and females, are killed per year in these traditional fisheries southwest of Kai Kecil during October-April (*in* Chan and Liew, 1996) (the Kai Islands take is assumed included in this estimate), and these takes are most likely continuing (C. Starbird, personal communication, 1998, *in* Clever Magazine, Issue No. 6).

As shown in Table III-3, since the early-to-mid 1980s, the number of female leatherback turtles nesting annually on the two primary beaches of Irian Jaya appear to be stable. However, given the current, serious threats to all life stages of the Indonesian leatherback populations, this trend may not be sustained and this population could collapse, similar to what occurred in Terengganu, Malaysia. As human populations in Indonesia increase, the need for meat and competition between the expanding human population and turtles for space increases, all leading to more direct takes of leatherback turtles or incidental take by local fisheries. There is no evidence to indicate that the preceding threats are not continuing today, as problems with nest destruction by feral pigs, beach erosion, and harvest of adults in local waters have been reported (Suarez *et al.*, unpublished report). In addition, local Indonesian villagers report dramatic declines in local sea turtle populations (Suarez, 1999); without adequate protection of nesting beaches, emerging hatchlings, and adults, this population will continue to decline.

Regarding the status of the Irian Jaya population of nesting leatherback turtles, Suarez *et al.* (*in press*) comment: "Given the high nest loss which has occurred along this coast for over thirty years it is not unlikely that this population may also suddenly collapse. Nesting activity must also continue to be monitored along this coast, and nest mortality must be minimized in order to prevent this population of leatherback turtles from declining in the future."

#### *Conclusion on status of eastern and western Pacific leatherback turtles*

Although quantitative data on human-caused mortality are scarce available information suggests that leatherback mortality on many nesting beaches remains at unsustainable levels (Tillman, 2000). In addition, except for elimination of fishing mortality in the now-defunct high-seas driftnet fisheries in the North and South Pacific, and reductions of effort in a few other fisheries, risks of mortality in fisheries generally have not been reduced.

Conservation efforts during the last few years at nesting beaches in Mexico and Costa Rica have led to increased survival of eggs, and therefore greater hatchling production per nesting female. This has the potential for increasing future recruitment if post-hatchling survival is not further reduced; however, since numbers of nests are so low, and post-hatchling and juvenile natural mortality are assumed to be high, this increase in hatchling production may only result in the addition of a few adults annually. In western Pacific populations, particularly Irian Jaya, nest destruction by beach erosion and feral pig

predation is widespread, and hatchling production is likely to be low relative to the numbers of nests laid. Overall, both eastern and western Pacific populations appear to have low female abundance as a result of legal harvest of eggs and nesting females, poaching, and incidental take in fisheries.

Representation in the various age classes of female leatherback turtles is most likely unbalanced as a result of losses of adult females, juveniles and eggs and sub-adults and adults as a result of on-going fisheries and the now-defunct high seas driftnet fisheries. Gaps in age structure may cause sudden collapse of nesting populations when age classes with few individuals recruit into the reproductive population as older individuals die or are removed.

### c. *Loggerhead Turtles*

The loggerhead turtle is listed as threatened under the ESA throughout its range, primarily due to direct take, incidental capture in various fisheries, and the alteration and destruction of its habitat. The loggerhead is categorized as Endangered, by the IUCN where taxa so classified are considered to be facing a very high risk of extinction in the wild in the near future. Loggerheads are a cosmopolitan species, found in temperate and subtropical waters and inhabiting pelagic waters, continental shelves, bays, estuaries and lagoons. In the Pacific Ocean, major nesting grounds are generally located in temperate and subtropical regions, with scattered nesting in the tropics ( *in* NMFS and USFWS, 1998c).

The loggerhead is characterized by a reddish brown, bony carapace, with a comparatively large head, up to 25 cm wide in some adults. Adults typically weigh between 80 and 150 kg, with average CCL measurements for adult females worldwide between 95-100 cm CCL (*in* Dodd, 1988) and adult males in Australia averaging around 97 cm CCL (Limpus, 1985, *in* Eckert, 1993). Juveniles found off California and Mexico measured between 20 and 80 cm (average 60 cm) in length (Bartlett, 1989, *in* Eckert, 1993). Skeletochronological age estimates and growth rates were derived from small loggerheads caught in the high-seas driftnet fishery. Loggerheads less than 20 cm were estimated to be 3 years or less, while those greater than 36 cm were estimated to be 6 years or more. Age-specific growth rates for the first 10 years were estimated to be 4.2 cm/year (Zug, *et al.*, 1995).

Nesting of loggerheads in the Pacific Basin are restricted to the western and southern region (Japan and Australia, primarily); there are no reported loggerhead nesting sites in the eastern or central Pacific. Upon reaching maturity, adult females migrate long distances from resident foraging grounds to their preferred nesting beaches. The average re-migration interval is between 2.6 and 3.5 years, in Queensland, Australia (*in* NMFS and USFWS, 1998c). Nesting is preceded by offshore courting, and individuals return faithfully to the same nesting area over many years. Clutch size averages 110 to 130 eggs, and one to six clutches of eggs are deposited during the nesting season (Dodd, 1988). Based on skeletochronological and mark-recapture studies, mean age at sexual maturity for loggerheads ranges between 25 to 35 years of age, depending on the stock (*in* Chaloupka and Musick, 1997), although Frazer *et al.* (1994 *in* NMFS and USFWS, 1998c) determined that maturity of loggerheads in Australia occurs between 34.3 and 37.4 years of age.

The transition from hatchling to young juvenile occurs in the open sea, and evidence is accumulating that

this part of the loggerhead life cycle may involve trans-Pacific developmental migration (Bowen, *et al.*, 1995). The size structure of loggerheads in coastal and nearshore waters of the eastern and western Pacific suggest that Pacific loggerheads have a pelagic stage similar to the Atlantic. This is supported by the fact that the high seas driftnet fishery, which operated in the Central North Pacific in the 1980s and early 1990s, incidentally caught juvenile loggerheads (mostly 40-70 cm in length) (Wetherall, *et al.*, 1993). In addition, large aggregations of mainly juveniles and subadult loggerheads, numbering in the thousands, are found off the southwestern coast of Baja California, over 10,000 km from the nearest significant nesting beaches (Pitman, 1990; Nichols, *et al.*, 2000). Genetic studies have shown these animals originate from Japanese nesting stock (Bowen *et al.*, 1995), and their presence reflects a migration pattern probably related to their feeding habits (Cruz, *et al.*, 1991, *in* Eckert, 1993). These loggerheads are primarily juveniles, although carapace length measurements indicate that some of them are 10 years old or older. Loggerheads tagged in Mexico and California with flipper and/or satellite transmitters have been monitored returning to Japanese waters (Resendiz, *et al.*, 1998a-b).

Tagging programs to study migration and movement of sea turtles provide evidence that loggerhead turtles are highly migratory and capable of trans-Pacific movement. Satellite telemetry studies show that loggerhead turtles tend to follow 17E and 20EC sea surface isotherms north of the Hawaiian islands (Polovina, *et al.*, 2000; Eckert, unpublished data). Relationships between other turtle species and sea surface temperatures have also been demonstrated, with most species preferring distinct thermal regimes (Stinson, 1984). After capture in the Hawaii-based longline fishery, six satellite transmitter-equipped loggerheads traveled westward along two convergent oceanic fronts, against prevailing currents and associated with a “cool” front characterized by sea surface temperature (17EC), surface chlorophyll and an eastward geostrophic current of about 4 centimeters/second (cm/sec). Three others were associated with a warmer front (20EC), lower chlorophyll levels, and an eastward geostrophic flow of about 7 cm/sec. This study supports a theory that fronts are important juvenile habitat (Polovina, *et al.*, 2000). Genetic analyses of 124 loggerheads caught in the Hawaii-based longline fishery indicated that the majority (nearly 100 percent) originated from Japanese nesting stock (P. Dutton, NMFS, personal communication, January, 2001). Loggerheads are not commonly found in U.S. Pacific waters, and there have been no documented strandings of loggerheads off the Hawaiian Islands in nearly 20 years (1982-1999 stranding data, G. Balazs, NMFS, personal communication, 2000).

For their first years of life, loggerheads forage in open ocean pelagic habitats. Both juvenile and subadult loggerheads feed on pelagic crustaceans, mollusks, fish, and algae. The large aggregations of juveniles off Baja California have been observed foraging on dense concentrations of the pelagic red crab, *Pleuronocodes planipes* (Pitman, 1990; Nichols, *et al.*, 2000). Data collected from stomach samples of turtles captured in North Pacific driftnets indicate a diet of gastropods (*Janthina* sp.), heteropods (*Carinaria* sp.), gooseneck barnacles (*Lepas* sp.), pelagic purple snails (*Janthina* sp.), medusae (*Vellela* sp.), and pyrosomas (tunicate zooids). Other common components include fish eggs, amphipods, and plastics (Parker, *et al.*, *in press*). These loggerheads in the north Pacific are opportunistic feeders that target items floating at or near the surface, and if high densities of prey are present, they will actively forage at depth (Parker, *et al.*, *in press*). As they age, some loggerheads begin to move into shallower waters, where, as adults, they forage over a variety of benthic hard- and

soft-bottom habitats (reviewed *in* Dodd, 1988). Subadults and adults are found in nearshore benthic habitats around southern Japan, in the East China Sea and the South China Sea (e.g. Philippines, Taiwan, and Viet Nam).

Studies of loggerhead diving behavior indicate varying mean depths and surface intervals, depending on whether they were located in shallow coastal areas (short surface intervals) or in deeper, offshore areas (longer surface intervals). Loggerheads appear to spend a longer portion of their dive time on the bottom (or suspended at depth), which may be related to foraging and refuge. Unlike the leatherback, to the loggerhead foraging in the benthos, bottom time may be more important than absolute depth (Eckert, *et al.*, 1989). The maximum recorded dive depth for a post-nesting female was 211-233 meters, while mean dive depths for both a post-nesting female and a subadult were 9-22 meters. Routine dive times for a post-nesting female were between 15 and 30 minutes, and for a subadult, between 19 and 30 minutes (Sakamoto, *et al.*, 1990 *in* Lutcavage and Lutz, 1997).

Loggerhead nesting in the Pacific basin is restricted to the western region, primarily Japan and Australia. In the western Pacific the only major nesting beaches are in the southern part of Japan (Dodd, 1988), but the population status of the loggerhead nesting colonies here and the surrounding region are less clear. Balazs and Wetherall (1991) speculated that 2,000 to 3,000 female loggerheads may nest annually in all of Japan; however, more recent data suggest that only approximately 1,000 female loggerhead turtles may nest there (Bolten *et al.* 1996). Nesting beach monitoring at Gamoda (Tokushima Prefecture) has been ongoing since 1954. Surveys at this site showed a marked decline in the number of nests between 1960 and the mid-1970s. Since then, the number of nests has fluctuated, but has been downward since 1985 (Bolten *et al.*, 1996). Monitoring on several other nesting beaches, surveyed since the mid-1970s, revealed increased nesting during the 1980s before declining during the early 1990s. Quantitative data on nesting levels since 1995 are unavailable, but are reported to show a continuing decline (Tillman, 2000). Nesting of loggerheads may also occur along the south China Sea, but it is a rare occurrence (Marquez, 1990, *in* Eckert, 1993).

In the south Pacific, Limpus (1982) reported an estimated 3,000 loggerheads nesting annually in Queensland, Australia during the late 1970s. However, long-term trend data from Queensland indicate a 50 percent decline in nesting by 1988-89, due to incidental mortality of turtles in the coastal prawn fishery. This decline is corroborated by studies of breeding females at adjacent feeding grounds (Limpus and Reimer, 1994). By 1997, the number of females nesting annually in Queensland was thought to be as low as 300 (1998 Draft Recovery Plan for Marine Turtles in Australia). Survey data are not available for other nesting assemblages in the south Pacific. Scattered nesting has also been reported on Papua New Guinea, New Zealand, Indonesia, and New Caledonia; however, population sizes on these islands have not been ascertained (NMFS and USFWS, 1998c).

There are no records of nesting loggerheads in the Hawaiian islands (Balazs, 1982), or in any of the islands of Guam, Palau, the Northern Mariana Islands (Thomas, 1989), the Federated States of Micronesia (Pritchard, 1982), or American Samoa (Tuato'o-Bartley, *et al.*, 1993). There are very few records of loggerheads nesting on any of the many islands of the central Pacific, and the species is considered rare or vagrant on islands in this region (NMFS and USFWS, 1998c).

As mentioned, aggregations of juvenile loggerheads off Baja California Mexico have been reported, although their status with regard to increasing or declining abundance has not been determined. NMFS and USFWS (1998c) report “foraging populations ... range from ‘thousands, if not tens of thousands’ (Pitman, 1990) to ‘at least 300,000 turtles’ (Bartlett, 1989). Extrapolating from 1988 offshore census data, Ramirez-Cruz et al. (1991) estimated approximately 4,000 turtles in March, with a maximum in July of nearly 10,000 turtles.”

Loggerhead mortality from human activities is not well-documented, except for estimates based on NMFS observer data in the Hawaii-based longline fishery and the CA/OR drift gillnet fishery and recent ongoing studies in Baja California, Mexico (Nichols, *et al.*, 2000; W. Nichols, University of Arizona, personal communication, 2000). A high mortality in the North Pacific high-seas driftnet fisheries of Japan, Republic of Korea, and Taiwan was estimated in the 1990s, but those fisheries no longer operate. Mortality of loggerheads in the East China Sea and other benthic habitats of this population are a concern and thought to be “high,” but have not been quantified (Kamezaki, personal communication, *in* Tillman, 2000).

#### d. *Olive Ridley Turtle*

Although the olive ridley is regarded as the most abundant sea turtle in the world, olive ridley populations on the Pacific coast of Mexico are listed as endangered under the ESA; all other populations are listed as threatened. The olive ridley is categorized as endangered by the IUCN, where taxa so classified are considered to be facing a very high risk of extinction in the wild in the near future (IUCN Red List, 2000). They are the smallest living sea turtle, with an adult carapace length between 60 and 70 cm, and rarely weighing over 50 kg. They are olive or grayish green above, with a greenish white underpart, and adults are moderately sexually dimorphic (NMFS and USFWS, 1998d).

Like leatherback turtles, most olive ridley turtles lead a primarily pelagic existence (Plotkin *et al.*, 1993), migrating throughout the Pacific, from their nesting grounds in Mexico and Central America to the north Pacific. While olive ridleys generally have a tropical range, with a distribution from Baja California, Mexico to Chile (Silva-Batiz, *et al.*, 1996), individuals do occasionally venture north, some as far as the Gulf of Alaska (Hodge and Wing, 2000). Surprisingly little is known of their oceanic distribution and critical foraging areas, despite being the most populous of north Pacific sea turtles. The post-nesting migration routes of olive ridleys tracked via satellite from Costa Rica traversed thousands of kilometers of deep oceanic waters, ranging from Mexico to Peru, and more than 3,000 kilometers out into the central Pacific (Plotkin, *et al.*, 1993). The turtles appeared to occupy a series of foraging areas geographically distributed over a very broad range within their oceanic habitat (Plotkin, *et al.*, 1994). The species appears to forage throughout the eastern tropical Pacific Ocean, often in large groups, or flotillas, and are occasionally found entangled in scraps of net or other floating debris. In a three year study of communities associated with floating objects in the eastern tropical Pacific, Arenas and Hall (1992) found sea turtles, present in 15 percent of observations and suggested that flotsam may provide the turtles with food, shelter, and/or orientation cues in an otherwise featureless landscape. Olive ridleys comprised the vast majority (75%) of these sea turtle sightings. Small crabs, barnacles and other marine life often reside on the debris and likely serve as food attractants to turtles. Thus, it is

possible that young turtles move offshore and occupy areas of surface current convergences to find food and shelter among aggregated floating objects until they are large enough to recruit to benthic feeding grounds of the adults. Olive ridleys feed on tunicates, salps, crustaceans, other invertebrates and small fish. Although they are generally thought to be surface feeders, olive ridleys have been caught in trawls at depths of 80-110 meters (NMFS and USFWS, 1998d), and a post-nesting female reportedly dove to a maximum depth of 290 meters. The average dive length for an adult female and adult male is reported to be 54.3 and 28.5 minutes, respectively (Plotkin, 1994, *in* Lutcavage and Lutz, 1997).

Olive ridley turtles begin to aggregate near the nesting beach two months before the nesting season, and most mating is generally assumed to occur in the vicinity of the nesting beaches, although copulating pairs have been reported over 100 km from the nearest nesting beach. Olive ridleys are considered to reach sexual maturity between 8 and 10 years of age, and approximately 3 percent of the number of hatchlings recruit to the reproductive population (Marquez, 1982 and Marquez, 1992, *in* Salazar, *et al.*, 1998). The mean clutch size for females nesting on Mexican beaches is 105.3 eggs, in Costa Rica, clutch size averages between 100 and 107 eggs (*in* NMFS and USFWS, 1998d). Females generally lay 1.6 clutches of eggs per season by Mexico (Salazar, *et al.*, 1998) and two clutches of eggs per season in Costa Rica (Eckert, 1993). Data on the remigration intervals of olive ridleys in the eastern Pacific are scarce; however, in the western Pacific (Orissa, India), females showed an annual mean remigration interval of 1.1 years. Reproductive span in females of this area was shown to be up to 21 years (Pandav and Kar, 2000).

Historically, an estimated 10 million olive ridleys inhabited the waters in the eastern Pacific off Mexico (Clifton, *et al.*, 1982 *in* NMFS and USFWS, 1998d). However, human-induced mortality led to declines in this population. Beginning in the 1960s, and lasting over the next 15 years, several million adult olive ridleys were harvested by Mexico for commercial trade with Europe and Japan. (NMFS and USFWS, 1998d). Although olive ridley meat is palatable, it was not widely sought after; its eggs, however, are considered a delicacy. Fisheries for olive ridley turtles were also established in Ecuador during the 1960s and 1970s to supply Europe with leather. (Green and Ortiz-Crespo, 1982).

In the eastern Pacific, nesting occurs all along the Mexico and Central American coast, with large nesting aggregations occurring at a few select beaches located in Mexico and Costa Rica. Few turtles nest as far north as southern Baja California, Mexico (Fritts, *et al.*, 1982) or as far south as Peru (Brown and Brown, 1982). A single olive ridley reportedly nested in 1985 on the island of Maui, Hawaii, but the eggs did not hatch (Balazs and Hau, 1986 *in* NMFS and USFWS, 1998d), and the event was most likely an anomaly. Where population densities are high enough, nesting takes place in synchronized aggregations known as *arribadas*. The largest known *arribadas* in the eastern Pacific are off the coast of Costa Rica (~475,000 - 650,000 females estimated nesting annually) and in southern Mexico (~800,000+ nests/year at La Escobilla, in Oaxaca (Millán, 2000).

The nationwide ban on commercial harvest of sea turtles in Mexico, enacted in 1990, has improved the situation for the olive ridley. Surveys of important olive ridley nesting beaches in Mexico indicate increasing numbers of nesting females in recent years (Marquez, *et al.*, 1995; Arenas, *et al.*, 2000).



Annual nesting at the principal beach, Escobilla Beach, Oaxaca, Mexico, averaged 138,000 nests prior to the ban, and since the ban on harvest in 1990, annual nesting has increased to an average of 525,000 nests (Salazar, *et al.*, in press). At a smaller olive ridley nesting beach in central Mexico, Playon de Mismalayo, nest and egg protection efforts have resulted in more hatchlings, but the population is still “seriously decremented and is threatened with extinction” (Silva-Batiz, *et al.*, 1996). Still, there is some discussion in Mexico that the species should be considered recovered (Arenas, *et al.*, 2000).

In Costa Rica, 25,000 to 50,000 olive ridleys nest at Playa Nancite and 450,000 to 600,000 turtles nest at Playa Ostional each year (NMFS and USFWS, 1998d). In an 11-year review of the nesting at Playa Ostional, (Ballesteros, *et al.*, 2000) report that the data on numbers of nests deposited is too limited for a statistically valid determination of a trend; however, there does appear to be a six-year decrease in the number of nesting turtles. At Playa Nancite, concern has been raised about the vulnerability of offshore aggregations of reproductive individuals to “trawlers, longliners, turtle fishermen, collisions with boats, and the rapidly developing tourist industry” (Kalb, *et al.*, 1996). The greatest single cause of olive ridley egg loss comes from the nesting activity of conspecifics on *arribada* beaches, where nesting turtles destroy eggs by inadvertently digging up previously laid nests or causing them to become contaminated by bacteria and other pathogens from rotting nests nearby. At a nesting site in Costa Rica, an estimated 0.2 percent of 11.5 million eggs laid during a single *arribada* produced hatchlings (in NMFS and USFWS, 1998d). In addition, some female olive ridleys nesting in Costa Rica have been found afflicted with the fibropapilloma disease (Aguirre, *et al.*, 1999).

At Playa La Flor, the second most important nesting beach for olive ridleys on Nicaragua, Ruiz (1994) documented 6 *arribadas* (defined as 50 or more females resting simultaneously). The main egg predators were domestic dogs and vultures (*Coragyps atratus* and *Cathartes aura*).

Although olive ridley *arribadas* in Orissa, India are among the largest such sites in the world, in the western Pacific, olive ridleys are not as well documented as in the eastern Pacific, nor do they appear to be recovering as well (with the exception of Orissa, India, only in recent years). There are a few sightings of olive ridleys from Japan, but no report of egg-laying. Similarly, there are no nesting records from China, Korea, the Philippines, or Taiwan. No information is available from Viet Nam or Kampuchea (in Eckert, 1993). In Thailand, olive ridleys occur along the southwest coast, on the Surin and Similan islands, and in the Andaman Sea. On Phra Thong Island, on the west coast of Thailand, the number of nesting turtles have declined markedly from 1979 to 1990. During the 1996-97 survey, only six olive ridley nests were recorded, and of these, half were poached, and one was predated by feral dogs. During the 1997-98 survey, only three nests were recorded. The main threats to turtles in Thailand include egg poaching, harvest and subsequent consumption or trade of adults or their parts (i.e. carapace), indirect capture in fishing gear, and loss of nesting beaches through development (Aureggi, *et al.*, 1999).

Indonesia and its associated waters also provides habitat for olive ridleys, and there are some recently documented nesting sites. On Jamursba-Medi beach, on the northern coast of Irian Jaya, 77 olive ridley nests were documented from May to October, 1999 (Teguh, 2000 in Putrawidjaja, 2000). However, as mentioned in the leatherback subsection, extensive hunting and egg collection, in addition

to rapid rural and urban development, have reduced nesting activities in this area. In Jayapura Bay, olive ridleys were often seen feeding, and in June, 1999, an estimated several hundred ridleys were observed nesting on Hamadi beach, despite heavy human population in the nearby area. Locals report daily trading and selling of sea turtles and their eggs in the local fish markets (Putrawidjaja, 2000). At Alas Purwo National Park, located at the eastern-most tip of East Java, olive ridley nesting was documented from 1992-96. Recorded nests were as follows: from September, 1993 to August, 1993, 101 nests; between March and October, 1995, 162 nests; and between April and June, 1996, 169 nests. From this limited data, no conclusions could be reached regarding population trends (Suwelo, 1999).

Olive ridleys nest on the eastern and western coasts of peninsular Malaysia; however, nesting has declined rapidly in the past decade. The highest density of nesting was reported to be in Terengganu, Malaysia, and at one time yielded 240,000 *eggs* (2,400 nests, with approximately 100 eggs per nest) (Siow and Moll, 1982, *in* Eckert, 1993)), while only 187 *nests* were reported from the area in 1990 (Eckert, 1993). In eastern Malaysia, olive ridleys nest very rarely in Sabah and only a few records are available from Sarak (*in* Eckert, 1993).

Olive ridleys are the most common species found along the east coast of India, migrating every winter to nest en-masse at three major rookeries in the state of Orissa, Gahirmatha, Robert Island, and Rushikulya (*in* Pandav and Choudhury, 1999). The Gahirmatha rookery, located along the northern coast of Orissa, hosts the largest known nesting concentration of olive ridleys. Unfortunately, uncontrolled mechanized fishing in areas of high sea turtle concentration, primarily illegally operated trawl fisheries, has resulted in large scale mortality of adults during the last two decades. Records of stranded sea turtles have been kept since 1993. Since that time, over 50,000 strandings of olive ridleys have been documented (*in* Shanker and Mohanty, 1999), and much of it is believed to be due to near-shore shrimp trawling. Fishing in coastal waters off Gahirmatha was restricted in 1993 and completely banned in 1997 with the formation of a marine sanctuary around the rookery. However, mortality due to shrimp trawling reached a record high of 13,575 ridleys during the 1997-98 season, and none of the approximately 3,000 trawlers operating off the Orissa coast use turtle excluder devices in their nets (Pandav and Choudhury, 1999), despite mandatory requirements passed in 1997. "Operation Kachhapa" was developed in the late 1990s to protect sea turtles and their habitat by enabling strict enforcement of the 5 km non-mechanized fishing zone limit, as well as putting forward efforts to monitor nestings and educate local inhabitants and fishermen (Shanker and Mohanty, 1999). However, shrimp boats continue to fish close to shore within this protected zone and continue to not use turtle excluder devices. Threats to these sea turtles also include artificial illumination and unsuitable beach conditions, including reduction in beach width due to erosion (Pandav and Choudhury, 1999).

According to Pandav and Choudhury (1999), the number of nesting females at Gahirmatha has declined in recent years, although after three years of low nestings, the 1998-99 season showed an increasing trend (Noronha, Environmental News Service, April 14, 1999), and the 1999-2000 season had the largest recorded number of olive ridleys nesting in 15 years (The Hindu, March 27, 2000; The Times of India, November 15, 2000). During the 1996-97 and 1997-98 seasons, there were no mass nestings of olive ridleys. During the 1998-99 nesting season, around 230,000 females nested during the

first arribada, lasting approximately a week (Pandav and Kar, 2000); unfortunately, 80% of the eggs were lost due to inundation and erosion (B. Pandav, personal communication, *in* Shanker and Mohanty, 1999). During 1999-2000, over 700,000 olive ridleys nested at Nasi islands and Babubali island, in the Gahirmatha coast. It is not known how many eggs and nests were lost to high winds and strong waves, predicted to cause erosion on the islands (The Hindu, March 27, 2000), and an estimated 6,000 turtles were killed during this period due to illegal mechanized trawlers and non-use of the prohibited turtle excluder devices (S. Sahoo, January, 2001 in [rediff.com](http://rediff.com)<sup>8</sup>).

There are no records of nesting on the unincorporated U.S. territories in the North Pacific. In the central Pacific, a single nesting was reported in September, 1985 on the island of Maui, Hawaii (*in* Eckert, 1993).

While olive ridleys generally have a tropical to subtropical range, individuals do occasionally venture north, some as far as the Gulf of Alaska. The post-nesting migration routes of olive ridleys, tracked via satellite from Costa Rica, traversed thousands of kilometers of deep oceanic waters ranging from Mexico to Peru and more than 3,000 kilometers out into the central Pacific (Plotkin *et al.* 1993). Stranding records since 1982 indicate that the olive ridley is the third most often stranded sea turtle in the Hawaiian Islands, averaging 2 per year (20 strandings from 1982-99) (G. Balazs, NMFS, personal communication, 2000).

Recent genetic information analyzed from 20 olive ridleys taken in the Hawaii-based longline fishery indicate that 60% of the turtles (n=12) originated from the eastern Pacific (Mexico and Costa Rica) and 40% of the turtles (n=8) were from the Indian and western Pacific rookeries (P. Dutton, NMFS, personal communication, January, 2001), indicating the animals from both sides of the Pacific converge in the north Pacific pelagic environment. An olive ridley taken in the CA/OR drift gillnet fishery originated from an eastern Pacific stock (i.e. Costa Rica or Mexico) (P.Dutton, NMFS, personal communication, September, 2000).

## 2. *Factors Affecting Sea Turtles in the Pacific Ocean*

Because impacts to sea turtles in the Pacific Ocean are generally non-discriminatory insofar as the different species are concerned, the following is a description of fisheries and non-fisheries-related threats to all sea turtles in the Pacific Ocean.

### a. *Fisheries impacts*

Very few fisheries in the Pacific Ocean are observed or monitored for bycatch. Rough estimates can be made of the impacts of coastal, offshore, and distant water fisheries on sea turtle populations in the Pacific Ocean by extrapolating data collected on fisheries with known effort that have been observed to incidentally take sea turtles. However, the point needs to be made that a straight extrapolation of this data

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<sup>8</sup><http://www.rediff.com/news/2001/jan/22oris.htm>

contains a large degree of uncertainty and variability. Sea turtles are not uniformly distributed, either by area, or by time of year. In addition, observer coverage of a fishery may be very low, observers may not always be randomly distributed on vessels, or they may be placed on vessels that use fishing strategy that may be uncharacteristic of the fleet. Also, surveys and logbooks may contain biased or incomplete information. Lastly, such take estimates are also hampered by a lack of data on pelagic distribution of sea turtles.

This section will summarize known fisheries that have been observed or reported to incidentally or intentionally take sea turtles. Appendix B provides a summary of current trends in fishing effort in the eastern and western Pacific Ocean, by year, and country. Estimates of total fishing effort are complicated by the fact that not all active vessels fish equivalent number of days per trip or annually, or use the same number of hooks, length of net, or mesh size, or have the same carrying capacity. However, even with minimum effort estimates, it is apparent that there is significant fishing effort in the Pacific Ocean for which NMFS has no bycatch information for sea turtles.

(i) North Pacific Driftnet Fisheries (before December 1992)

Foreign high-seas driftnet fishing in the north Pacific Ocean for squid, tuna and billfish ended with a United Nations moratorium in December, 1992. Except for observer data collected in 1990-1991, there is virtually no information on the incidental take of sea turtle species by the driftnet fisheries prior to the moratorium. Tables 1 and 2 in Appendix B provide a summary of the number of active Japanese, Korean, and Taiwanese vessels fishing mainly for tuna in the Central Western Pacific Ocean from 1990-99.

The high seas squid driftnet fishery in the North Pacific was observed in Japan, Korea, and Taiwan, while the large-mesh fisheries targeting tuna and billfish were observed in the Japanese fleet (1990-91) and the Taiwanese fleet (1990). A combination of observer data and fleet effort statistics indicate that 4,373 turtles, mostly loggerheads and leatherback turtles, were entangled by the combined fleets of Japan, Korea and Taiwan during June, 1990 through May, 1991, when all fleets were monitored (Table III-4). Of these incidental entanglements, an estimated 1,011 turtles were killed (77 percent survival rate).

**Table III-4. Estimated annual bycatch and mortality of sea turtles in the North Pacific high-seas driftnet fishery for squid, tuna & billfish in 1990-91 (Wetherall, 1997).**

Species	Estimated Annual Take	Estimated Annual Mortality
green	378	93
leatherback	1,002	111
loggerhead	2,986	805
<b>TOTAL</b>	4,366	1,009

Data on size composition of the turtles caught in the high-seas driftnet fisheries were also collected by

observers. Green turtles and the majority of loggerheads measured by observers were immature, and most of the actual measured leatherback turtles were immature, although the size of leatherback turtles that were too large to bring on board were only estimated, and are therefore unreliable (Wetherall, 1997).

These rough mortality estimates for a single fishing season provide only a narrow glimpse of the impacts of the driftnet fishery on sea turtles, and a full assessment of impacts would consider the turtle mortality generated by the driftnet fleets over their entire history and geographical range. Unfortunately, comprehensive data are lacking, but the observer data does indicate the possible magnitude of turtle mortality given the best information available. Wetherall *et al.* (1993) speculate that “the minimum total turtle mortality in the North Pacific high-seas driftnet fisheries may have been on the order of 2,500 turtles per year during the late 1980s. The actual mortality was probably greater than this, but less than the estimated total driftnet bycatch of perhaps 9,000 turtles per year. Based on 1990 observer data, most of the mortalities would have been loggerheads taken in the Japanese and Taiwanese large-mesh fisheries.”

While a comprehensive, quantitative assessment of the impacts of the North Pacific driftnet fishery on turtles is impossible without a better understanding of turtle population abundance, stock origins, exploitation history and population dynamics, it is likely that the mortality inflicted by the driftnet fisheries in 1990 and in prior years was significant (Wetherall *et al.* 1993), and the effects may still be evident in sea turtle populations today. The high mortality of juvenile, pre-reproductive adults, and reproductive adults in the high-seas driftnet fishery has probably altered the current age structure (especially if certain age groups were more vulnerable to driftnet fisheries) and therefore diminished or limited the reproductive potential of affected sea turtle populations.

## (ii) Japanese tuna longliners in the Western Pacific Ocean and South China Sea

Based on turtle sightings and capture rates reported in a survey of fisheries research and training vessels and extrapolated to total longline fleet effort by the Japanese fleet in 1978, Nishimura and Nakahigashi (1990) estimated that 21,200 turtles, including greens, leatherback turtles, loggerheads, olive ridleys and hawksbills, were captured annually by Japanese tuna longliners in the Western Pacific and South China Sea, with a reported mortality of approximately 12,300 turtles per year. Using commercial tuna longline logbooks, research vessel data and questionnaires, Nishimura and Nakahigashi (1990) estimated that for every 10,000 hooks in the Western Pacific and South China Sea, one turtle is captured, with a mortality rate of 42 percent. Although species-specific information is not available, vessels reported sightings of turtles in locations which overlap with commercial fishing grounds in the following proportions: loggerhead - 36 percent, green turtle - 19 percent, hawksbill - 10.3 percent, olive ridley - 1.7 percent, leatherback - 13.7 percent, and unknown - 19 percent.

Caution should be used in interpreting the results of Nishimura and Nakahigashi (1990), including estimates of sea turtle take rate (per thousand hooks) and resultant mortality rate, and estimates of annual take by the fishery, for the following reasons: (1) the data collected was based on observations by training and research vessels, logbooks and a questionnaire (i.e. hypothetical), and do not represent

actual, substantiated logged or observed catch of sea turtles by the fishery; (2) the authors assumed that turtles were distributed homogeneously; and (3) the authors used only one year (1978) to estimate total effort and distribution of the Japanese tuna longline fleet. Although the data and analyses provided by Nishimura and Nakahigashi (1990) are conjectural, longliners fishing in the Pacific have had, and (with the current level of effort) probably continue to have significant impacts on sea turtle populations. Unfortunately, current bycatch information is not available for these fisheries, and NMFS is unaware of any follow-up studies since 1990. Future investigations into the level of sea turtle bycatch in these fisheries would allow a more complete assessment of cumulative effects on pelagic sea turtles in the Pacific Ocean.

(iii) South American fisheries

*Chile*

Although data on the incidental take of sea turtles in the Chilean swordfish fisheries are sparse, both green and leatherback turtles have been confirmed taken and killed, and olive ridleys and loggerheads may also be taken incidentally by the fishery (Weidner and Serrano, 1997). As described further in Appendix B, the Chilean swordfish fishery is comprised primarily of artisanal fishermen, averaging 500 boats (mainly driftnetters) from 1989 to 1991, and decreasing in numbers after 1991. Since 1991, approximately 20 large industrial (i.e. commercial) boats have fished swordfish in Chile, the effort is comprised of gillnets (27%), pelagic longliners (72%) and boats that switch gear. Effort by the artisanal fishery (including the driftnet fleet) increased from 5,265 days-at-sea in 1987 to 41,315 days-at-sea in 1994 (Barbieri, *et al.*, 1998).

Adult female leatherback turtles tagged in Mexico have been taken in Chilean waters by gillnet *and* purse seine fisheries (Marquez and Villanueva, 1993). In addition, data were recorded opportunistically from the artisanal swordfish fishery (driftnetters primarily) for a single port (San Antonio) over a two year period. This partial record documented leatherback captures and sightings totaling 9 in 1988 and 21 in 1989. A rough estimate of 250 leatherback takes per year without differentiating between kills and total takes for vessels operating out of San Antonio was provided (Frazier and Brito Montero, 1990). A more recent estimated annual take of 500 leatherback turtles was provided by Montero (personal communication, 1997, *in* Eckert, 1997) which was not unreasonable, given the nearly ten-fold increase in fishing effort from 1987 to 1994.<sup>9</sup> As shown in Table III-5, the take of sea turtles by the artisanal driftnet fishery in the late 1980s appeared to be comprised primarily of leatherback turtles.

Effort by the artisanal driftnet fishery for swordfish appears to be relatively constant through 1996, as shown in Table III-6. Given the total sea turtle take estimate from the 1988-89 season, and combining

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<sup>9</sup>Based on all information from Chile and Peru, Eckert (1997) estimated that a minimum of 2,000 leatherback turtles are killed annually by Peruvian and Chilean swordfish operations, representing a major source of mortality for leatherback turtles originating from and returning to nesting beaches in Costa Rica and Mexico. Because swordfish fishing effort has declined significantly since the early 1990s, incidental take has most likely declined as well, although the current estimate is unknown.

it with the total effort (days-at-sea) data from 1988-1996, and assuming effort was constant and in the same general area during all years, a simple calculation can be made to estimate the incidental take of turtles by the Chilean artisanal driftnet fishery for swordfish during subsequent years (third column in Table III-6). Turtles reportedly began appearing in Chilean markets in 1987, just as the swordfish driftnet fishery was expanding, and Chilean observers have reported occasional individual sets with leatherback mortalities of from 3-13 (*in* Weidner and Serrano, 1997). Assuming the current artisanal driftnet fishing effort is equivalent to 1996 and assuming the proportion of species taken is equivalent to data collected from the 1988-89 fishing season, this fishery would currently take an estimated 39 greens, 76 leatherback turtles, 4 loggerheads, and 29 olive ridleys annually.

During 1996, there was a substantial expansion of Chilean longline fishing in offshore areas, but as there has been no collection of data on this fishery as of 1997 (Weidner and Serrano, 1997), the anticipated effects on sea turtle stocks as a result in this change in fishing strategy are not known. Since effort for swordfish in the Chilean fishery or throughout the Pacific has declined significantly overall since 1994 (as a result of concerns about overfishing swordfish stocks) the bycatch of sea turtles in this fishery has likely declined as well, although the extent of this decrease is currently unknown. There is very little information on lethal and non-lethal incidental catch per unit effort. In addition to the swordfish fishery, Chile also has a substantial purse seine fleet, which has recently shifted from a reliance on coastal anchovy and sardines to a substantial take of jack mackerel further offshore, where turtle interactions may be more common (Weidner and Serrano, 1997). The extent of the impact of the Chilean purse seine fishery on sea turtles is unknown.

**Table III-5. Chile – turtle bycatch of artisanal driftnet fishermen, 1988-89.**

Species	Number	Percentage of Total
Green turtle	42	28%
Leatherback	82	55%
Loggerhead	5	3%
Olive ridley	21	14%
Total	150	100%

Source: José Brito-Montero, personal communication, 3/3/97, *in* Weidner and Serrano, 1997

**Table III-6. Chile - artisanal (driftnet) swordfish effort, by year, from 1989-1996 and calculated (not actual or known) turtle take [note assumptions used in this Opinion].**

Year	Effort (Days-at-sea)	Calculated Turtle Take*
1989	7,579	150*
1990	6,226	123
1991	11,450	227
1992	11,209	222
1993	10,755	213
1994	8,393	166
1995	8,152	161
1996	7,041	139

\*Calculated turtle take was estimated by comparing effort for 1989 (7,579 days-at-sea) and a known turtle take of 150 (1988-89 season) with subsequent years for which effort was known, but turtle take is not known.

\*\*Estimated take of turtles by Brito-Montero, for the 1988-89 season, and assuming 1989 data is equivalent in effort to 1988-89 effort, for the purpose of comparing year-to-year calculations of estimated turtle take. Source: Weidner and Serrano, 1997.

### *Colombia*

A description of known Colombian commercial fisheries is provided in Appendix B and summarized in Table 5 of the Appendix. No information is available on the sea turtle bycatch levels in the shrimp trawl fisheries and other fisheries operating out of Colombia. However, a turtle excluder device program has been initiated in the shrimp trawl fishery to reduce incidental catch. Artisanal fisheries in the past targeted turtles (Weidner and Serrano, 1997); however, no recent information on directed take is available.

### *Ecuador*

Appendix B contains a description of known current commercial and artisanal fisheries in Ecuador. Unfortunately, the composition of turtle species incidentally taken by Ecuadorian commercial and artisanal fisheries is unavailable. Prior to a ban on the commercial harvest for olive ridleys in 1986, artisanal fishermen prosecuted a directed turtle fishery as well as taking them incidentally. During 1985 and 1986, 124 and 715 metric tons of turtles, respectively, were reportedly taken (Table III-7). In 1990, the Ecuadorian government permanently ended the directed fishery, prohibiting the catch as well as domestic and export marketing. Incidental catches of sea turtles by tuna and swordfish longliners are reportedly very rare, but they do occur, and Ecuadorian authorities have seized turtle skins from Japanese longliners (*in* Weidner and Serrano, 1997).

### *Peru*

Appendix B contains a description of known domestic and foreign fisheries in Peru. Peruvian



commercial longline fleets have had limited success in fishing for swordfish, so there is probably very little incidental catch of sea turtles in this fishery. Peruvian artisanal fishermen, however, also target fish species normally taken in commercial longline fisheries (especially shark) and have been more successful than the commercial longline fleet, so more turtles may be caught incidental to these artisanal fisheries. Foreign longline fleets are also active and extensive off Peru and the bycatch of sea turtles in these foreign fisheries has been considered significant (Weidner and Serrano, 1997).

Peru conducted directed commercial turtle harvests throughout the 1980s, and, as recently as 1990, over 100 metric tons of turtles were taken (Table III-7) (FAO, Yearbook of Fishery Statistics, 1994, *in* Weidner and Serrano, 1997). Species-specific information was not available. Based on a sighting of 167 leatherback carapaces in a canyon near the port of Pucusana in 1978, Brown and Brown (1982) estimated a minimum of 200 leatherback turtles killed per year at that time. Furthermore, central Peru was known to have had the largest leatherback fishery in the world, taking what appeared to be adults and subadults, thus representing a considerable number of reproductive and near reproductive individuals (*in* Brown and Brown, 1982). The Ministerio de Pesqueria (MIPE), which is the Peruvian agency responsible for fisheries, prohibited the taking of all leatherback turtles and green turtles less than or equal to 80 cm in length through a resolution in January, 1977, although observers report that regulations are rarely enforced. Other species were not protected and were still unprotected as of 1989, although catches appear to have declined to negligible levels (Weidner and Serrano, 1997). Specific take levels remain unknown.

**Table III-7. Ecuador and Peru - turtle catch in metric tons, 1985-95.**

Year	Catch - Ecuador (metric tons)	Catch - Peru (metric tons)
1985	124	36
1986	715	9
1987	—	305
1988	—	32
1989	—	79
1990	—	101
1991	—	9
1992	—	30
1993	—	28
1994	—	6
1995	10*	4*
Source: FAO, Yearbook of Fishery Statistics, 1994, <i>in</i> Weidner and Serrano (1997) *1995 data would not be found in the above source, yet Weidner and Serrano (1997) provide data for this year.		

(iv) Distant Water Fishing Nations Longline Fishing in the EEZ around the Federated States of Micronesia

Heberer (1997) summarized the results of 51 distant-water fishing nation (DWFN) longline trips observed by Micronesian Maritime Authority fisheries observers from 1993 through 1995. Vessels from China, Taiwan, and Japan captured a total of 34 sea turtles. These turtles were reported as 15 olive ridleys, 8 green turtles, and 11 unidentified sea turtles. Thirty of the 34 turtles were released alive and the remainder were dead when landed (11.8% mortality rate). Data on hooking location or entanglement was not reported, nor was the condition of each turtle by species.

The Micronesia Fisheries Authority (previously Micronesian Maritime Authority) places observers aboard distant water fishing vessels fishing by longline in their EEZ. Table III-8 shows the observed catch of sea turtles by these vessels from January 1, 1990 through December, 2000. While the overall data set represents a significant amount of effort - 971 sets and 1,272,000 hooks observed over a 10 year period, the rate of observer coverage is extremely low. From 1990 through 1997, observer coverage ranged from 1 to 3%.

<b>Table III-8. Observed captures of sea turtles aboard distant water longline vessels, January 1990 through December 2000. Source: Micronesian Fisheries Authority</b>			
<b>Species</b>	<b>Number</b>	<b>Condition</b>	
		<b>% Alive</b>	<b>% Dead</b>
Green	4	100	0
Hawksbill	1	100	0
Loggerhead	1	100	0
Olive ridley	8	100	0
Unidentified turtle	33	79	21
Total	47		

The information presented above is from two separate data sets, which may not have been coordinated. The study done by Heberer (1997) utilized observers specifically trained and directed to record bycatch information, whereas observers in this fishery typically prioritize the collection of target catch data over bycatch information. This information represents the best available information on bycatch in this fishery. Appendix B provides additional information on fishing effort. However, the above data cannot be compared or used to extrapolate expected rates of turtle bycatch based on small sample sizes, low rates of observer coverage, and prioritization of catch data.

(v) U.S. tuna purse seine fishery in the eastern tropical Pacific Ocean (ETP)

The vast majority of the U.S. western and central Pacific purse seine activity occurs in the highly productive fishing grounds of the equatorial western Pacific (principally in the EEZs surrounding Papua New Guinea, the Federated States of Micronesia and Kiribati) under a multilateral agreement entitled *Treaty on Fisheries Between the Governments of Certain Pacific Island States and the Government of the United States of America* or the South Pacific Tuna Treaty (SPTT). The treaty was signed by the United States and 16 Pacific Island Parties belonging to the Forum Fisheries Agency (FFA), and provides U.S. tuna purse seiners access to tunas in a 25.9 million km<sup>2</sup> area of the central-western Pacific Ocean in exchange for fishing fees and adherence to rules related to closed area, etc (Coan, *et al.*, 1997). The treaty was renegotiated in 1992 for an additional 10 years.

Between 1988 and 1999, the number of licensed U.S. tuna purse seiners ranged from 35 to 51, although only between 31 and 49 vessels fished during those years. Between 71 and 241 trips were made during each calendar year (Coan, *et al.*, 2000), and most of the fishing was conducted in the equatorial belt, extending from around 155EW to 140EE longitude, the traditional fishing zone for the U.S. fleet (Coan, *et al.*, 1997). The U.S. fleet primarily lands their catch in American Samoa (Coan, *et al.*, 1997, 2000). From 1988 to 1995, the fleet primarily set on free-swimming school sets and less on log sets; however, beginning in 1996, sets were increasingly made on floating aggregation devices (FADs), and in 1999, nearly 100 % of sets were on FADs (Coan, *et al.*, 2000). Because turtles tend to congregate around floating objects in the open ocean, this change in fishing strategy may increase the likelihood of sea turtle interactions.

The number of large (>400 short tons carrying capacity) ETP tuna purse seine vessels has remained steady since 1992, varying between 5 and 7 vessels, and the number of smaller (#400 st) vessels has also remained steady, averaging 18 vessels between 1993 and 1997 (NMFS, 1998b). Although all large tuna purse seine vessels fishing in the ETP for tuna have been required to carry observers since 1989 (100 percent coverage), smaller purse seine vessels are not required to carry observers. Most smaller tuna vessels fishing off southern California fish on tuna schools because the vessels are old, slow, and lack the resources (e.g. helicopters) needed to place and find floating objects (B. Jacobson, NMFS, personal communication, 1999). Based on observer data from the large vessels, the chances of incidentally capturing a sea turtle during a school set are much less than incidentally capturing a sea turtle during floating object sets; therefore, the incidental take of sea turtles by the small vessel fleet is likely to be less than that of the larger purse seine vessels. However, with no observer coverage, data on sea turtle bycatch are not available for the small tuna purse seine vessels in the ETP.

The U.S. fleet is required to take Fisheries Forum Agency (FFA) observers on a minimum of 20 percent of their fishing trips, and captains are responsible for recording catch and bycatch data in logbooks. Logbooks are verified by observers, if possible, and are sent to the FFA no later than 14 days after returning to port (K. Staisch, FFA, personal communication, February, 2001). Between 1997 and 1999, there was approximately 20-23% observer coverage (Forum Fisheries Agency, 1998; A. Coan, personal communication, February, 2001). Collecting data on target species (i.e. tuna) is a priority for observers; however, if possible, and when time permits, observers do collect bycatch data.

Observers receive limited training on sea turtle identification and are trained to look for tags, but they do not collect information on length or take biopsies, as the turtles are generally released immediately from the net. The incidental catch of sea turtles is a “rare occurrence,” and any turtles observed taken have been released alive. Purse seine techniques normally allow turtles to surface for air during the pursing period, and based on observer reports, any turtles caught in nets are usually released as soon as possible. In addition, there have been no reports of turtles caught in the power block (K. Staisch, FFA, personal communication, February, 2001).

**Table III-9. Sea turtle interactions by U.S. tuna purse seine fleet (1990 - 1997) - large vessels only [Note: there is some discrepancy between the numbers in the two parts of the table because previously dead turtles were not included in species estimates and hawksbill turtles were not included in the top part of the table and not accounted for it in the lower part]**

<b>Set Summary / by calendar year 1/1 - 12/30</b>									
<b>Cruise Year</b>	<b>1990<sup>1</sup></b>	<b>1991</b>	<b>1992</b>	<b>1993</b>	<b>1994</b>	<b>1995</b>	<b>1996</b>	<b>1997</b>	<b>Total</b>
<b>Number of sea turtles taken (mortality in parentheses) by species<sup>2</sup></b>									
<b>Annual Average</b>									
Olive ridley	113(2)	104	132	133(1)	69	69(1)	45(1)	95(1)	<b>96</b>
Green turtle	4	8	21	35	28	29	17	11	<b>19</b>
Leatherback	3	0	0	2	1	0	0	0	<b>0.8</b>
Loggerhead	0	1	0	0	3	0	0	2	<b>0.8</b>
Unidentified	36	37	25(1)	21	19	3	25	8	<b>22</b>
<b>Totals</b>	<b>156</b>	<b>150</b>	<b>178</b>	<b>191</b>	<b>120</b>	<b>101</b>	<b>87</b>	<b>116</b>	<b>137</b>
<b>Condition of sea turtle when released (injury/mortality due to set)</b>									
<b>Annual Average</b>									
Prev. dead	0	0	2	1	4	2	0	2	<b>1.4</b>
Released unharmed	126	137	168	181	115	92	73	110	<b>127</b>
Released slightly injured	13	5	7	1	3	6	5	2	<b>5.3</b>
Kill accidentally	2	0	1	1	0	1	1	1	<b>0.9</b>
Escaped net	11	5	3	6	2	0	7	3	<b>4.7</b>
Other/unknown	3	3	0	2	0	4	1	2	<b>1.9</b>
<b>Totals</b>	<b>156</b>	<b>150</b>	<b>181</b>	<b>192</b>	<b>124</b>	<b>105</b>	<b>87</b>	<b>120</b>	<b>141.1</b>

<sup>1</sup>First year of sea turtle data collection, did not began until 3/20. Summary reflects cruises from 3/20/90 - 12/30/90, when data was collected. 1,629 sets out of 1,814 for 1990 were observed for sea turtles.

<sup>2</sup>Mortalities are a subset of total incidental take.

In addition to collecting tuna life history and marine mammal and bycatch data during a set, observers on large U.S. purse seiners in the ETP complete a sea turtle life history form when a sea turtle is taken in a set (i.e., sea turtle was captured or at any time entangled in the net). Table III-9 shows sea turtle interactions in the large U.S. tuna purse seine fleet from 1990 to 1997. Data for 1998 and most of 1999 has not been entered into a database and is therefore currently unavailable. The 1990-1997 data include 174 turtles taken in the fishery that were not identified to species, although only 1 of these unidentified turtles is listed as accidentally killed (as discussed earlier, these estimates may underestimate the number of sea turtles killed in the fishery because some turtles that were lethargic when they were released, which were considered “alive” when they were released, probably died from their injuries subsequent to their release). Most of unidentified sea turtles probably never came on board, but escaped after being encircled or captured, and the observer was not close enough to identify the turtle as it swam away. Assuming that these unidentified turtle interactions occurred in the same proportions as the identified sea turtle interactions, these 174 turtles would most likely be comprised of 143 olive ridleys, 28 green turtles, and 1 to 3 leatherback, hawksbill or loggerhead turtles, in unknown proportion. It is likely that most of these 174 unidentified turtles were uninjured by their capture or encirclement if they did release themselves from the net and swim away.

In its December 8, 1999, biological opinion on the effects of the interim final rule for the continued authorization of the ETP U.S. tuna purse seine fishery on listed species, NMFS estimated the maximum annual incidental takes and mortalities of sea turtles for 2000-2010: green - 35 taken, 2 killed; leatherback turtles - 2 taken, 1 killed every 10 years; loggerheads - 3 taken, 1 killed every 7 years; olive ridleys - 133 taken, 7 killed (NMFS, 1999).

(vi) Foreign tuna purse seine fishery in the ETP

The international fleet represents the majority of the fishing effort and carrying capacity in the ETP tuna fishery, with most of the total capacity consisting of purse seiners greater than 400 st. These large vessels comprised about 87 percent of the total fishing capacity operating in the ETP in 1996 (IATTC, 1998). An average of 107 foreign vessels with a carrying capacity greater than 400 st fished in the ETP during 1993 to 1997. In addition to these larger vessels, the foreign fleet contains smaller vessels less than 400 st that target tuna in the ETP. From 1993 to 1997, an average of 63 foreign vessels ranging from 45 to 400 st carrying capacity fished in the ETP each year.

Data from observers on both U.S. and foreign tuna purse seine vessels have been gathered collectively by the IATTC since the early 1990s (Table III-10; data are in addition to Table III-9). The most recent data from the IATTC indicate that an average of 172 sea turtles per year were killed by vessels over 400 st in the entire ETP purse seine fishery (U.S. included) from 1993-97 (IATTC, 1999).

The 1993-1997 data indicate that 168 turtles killed by the entire tuna purse seine fishery were “unidentified,” although the reasons for this were not given. Assuming that these unidentified turtle mortalities occurred in the same proportions as the identified turtle mortalities, these 168 turtles would be 140 olive ridleys, 20 green turtles, 7 loggerhead turtles and one would be either a leatherback or hawksbill.

Table III-10. Estimated sea turtle mortality by species for the entire ETP tuna purse seine fishery (U.S. and foreign) from 1993-1997 <sup>1</sup>					
Species/Year	1993	1994	1995	1996	1997
Olive ridley	197	103	94	83	99
Loggerhead	5	10	2	3	7
Green/black	39	8	12	7	19
Leatherback	0	0	0	1	0
Unidentified	46	36	32	29	25
<b>TOTAL</b>	287	157	140	123	150

<sup>1</sup> (M. Hall, IATTC, personal communication, 1999)

(vii) Mexican (Baja California) fisheries and direct harvest

Based on a combination of analyses of stranding data, tag-recapture studies and extensive interviews, all carried out between 1994 and 1999, Nichols (University of Arizona, personal communication, October 2000) has conservative estimates of the annual take of green turtles and loggerhead turtles by various fisheries and through direct harvest in the Baja California, Mexico region. Nichols and his affiliates estimated the annual mortality of green turtles in this region to be *greater* than 7,800 turtles, impacting both immature and adult turtles. Mortality of loggerhead turtles, based on stranding and harvest rates, is estimated at 1,950 annually, and affects primarily immature size classes. The primary causes for mortality are the incidental take in a variety of fishing gears and direct harvest for consumption and [illegal] trade.

Based on stranding patterns, Nichols, *et al.* (2000) speculate that mortality of loggerheads due to local fishing in Baja California may primarily be due to a net-based fishery. None of the stranded turtles showed signs of hooking; therefore the halibut fishery, which reports regular loggerhead bycatch and coincides with the movement of pelagic red crab into the shallower continental shelf, may interact with loggerheads as they enter coastal waters in the spring and summer.

(viii) California/Oregon drift gillnet fishery

The California/Oregon (CA/OR) drift gillnet fishery targets swordfish and thresher shark. The fishery has been observed by NMFS since July 1990, and observer coverage has ranged from 4.4 percent in 1990 to an estimated 21.0 percent in 1999. Between July 1990 and December 31, 1999, NMFS has observed 5,529 sets (NMFS unpublished data). The fishery occurs primarily within 200 nautical miles (nm) of the California coastline and to a lesser extent off the coast of Oregon. Under California state regulations, the fishery is restricted to waters outside 200 nm from February 1 through April 30 and outside 75 nm from May 1 through August 14. Fishing is allowed inside 75 nm from August 15 through January 31. Because of these restrictions, the fishery is not active during February, March, and April.

In addition, very little fishing effort occurs during the months of May, June, and July since CA/OR drift gillnet vessels targeting swordfish tend to set on warm ocean water temperature breaks which don't appear along the California coast until late summer. Currently, approximately 90 percent of the fishing effort occurs between August 15 and December 31. On average, about 9 percent of the fishing effort occurs during the month of January, 0 percent occurs February through April, and slightly more than 1 percent occurs between May 1 and August 14 (California Department of Fish and Game, unpublished data).

Fishers use nets constructed from 3-strand twisted nylon, tied to form meshes. The meshes range from 16 to 22 inches stretched, and average 19 inches stretched. Although termed "gillnets," the nets actually entangle fish, rather than trap them by the gills. Net length ranges from 750 to 1000 fathoms, averaging 960 fathoms. The top of the net is attached to a float line by hanging lines laced through several meshes and tied at intervals of 8 to 24 inches. The number of meshes per hanging determines the slack or tautness of the net. The bottom of the net is attached to a weighted lead line. The number of meshes between the float line and the lead line determines the depth of the net, which ranges from 100 to 150 meshes. The depth at which the float line is suspended in the water column is determined by the length of the buoy line (extender length). Nets are often set perpendicular to currents, or across temperature, salinity, or turbidity fronts. Nets are typically set in the evening, allowed to soak overnight, then retrieved in the morning. The average soak time is 10.5 hours (NMFS 1997b). The vessel remains attached to one end of the net during the soak period, drifting with the net.

The CA/OR drift gillnet fishery has been subject to the Pacific Offshore Cetacean Take Reduction Plan (PCTRP) since October 1997 (62 FR 51805). The PCTRP requires that nets be fished at a minimum depth of 36 feet below the water surface, that acoustic warning devices ("pingers") be used during all sets, and that skipper workshops be held to educate fishers about the take reduction plan requirements and solicit input on additional ways to possibly reduce marine mammal take. Based on a comparison of observer data collected prior to and since the implementation of the PCTRP, there does not appear to be a significant difference in sea turtle entanglement rates, although interactions are rare events in this fishery.

Green and olive ridley turtles are rarely taken by the CA/OR drift gillnet fishery; in fact, only one green and one olive ridley turtle have been observed since NMFS began observing the fishery in 1990. Both of these observed takes occurred in 1999. The green turtle was returned dead and the olive ridley was released alive. In addition, there have been 23 leatherback turtles observed taken by this fishery since 1990. Almost all of these interactions occurred north of Point Conception (34° 25' N), and 78% of these interactions occurred during the months of August, September, and October with the majority of the interactions occurring during October (61%). There have been 14 loggerhead sea turtle interactions observed in the CA/OR drift gillnet fishery. All of these interactions were south of Point Conception and occurred during El Niño events. Table III-11 shows the annual estimated mortality of sea turtles incidentally taken by the CA/OR drift gillnet fishery, based on extrapolated observer data. Animals released alive or injured are not included in the table.

**Table III-11. Estimated mortality (and coefficients of variation) of sea turtles by the California/Oregon drift gillnet fishery based on observer data.**

Species	1990 <sup>1</sup>	1991 <sup>1</sup>	1992 <sup>1</sup>	1993 <sup>1</sup>	1994 <sup>1</sup>	1995 <sup>1</sup>	1996 <sup>2</sup>	1997 <sup>3</sup>	1998 <sup>3</sup>	1999 <sup>4</sup>
Green	0	0	0	0	0	0	0	0	0	5 (0.90)
Loggerhead	0	0	7 (0.93)	0	0	0	0	6 (0.95)	5 (0.89)	0
Leatherback	23 (0.97)	0	15 (0.65)	15 (0.66)	0	26 (0.55)	24 (0.64)	7 (0.95)	0	0
Olive Ridley	0	0	0	0	0	0	0	0	0	0
Unidentified Turtle	0	0		7 (0.93)	0	0	0	0	0	0

<sup>1</sup> Julian and Beeson, 1998.

<sup>2</sup> Julian 1997.

<sup>3</sup> Cameron and Forney, 1999.

<sup>4</sup> Cameron and Forney, 2000.

On October 23, 2000, NMFS issued a biological opinion on the issuance of a permit under section 101(a)(5)(E) of the MMPA for the incidental taking of marine mammal species listed under the ESA during commercial fishing operations. After reviewing the available scientific and commercial data, current status of Pacific leatherback and loggerhead sea turtles, the environmental baseline for the action area, the effects of the proposed action and the cumulative effects, the opinion found that the issuance of section 101(a)(5)(E) permits and the associated continued operation of the CA/OR drift gillnet fishery, as regulated under the PCTRP, was likely to jeopardize the continued existence of Pacific leatherback and loggerhead sea turtles. Based on this opinion, NMFS shall implement regulations by August 1, 2001, that will eliminate drift gillnet fishing effort during the months of August, September, and October north of Point Conception to reduce the likelihood of interactions with leatherback turtles. In addition, fishing effort south of Point Conception will be eliminated during El Niño events in August and January to reduce the likelihood of an interaction with loggerhead turtles. As a result of these closures, NMFS expects there to be at least a 78% reduction in the overall leatherback interactions and 92% reduction in loggerhead interactions.

**b. *Other impacts***

Threats to sea turtles vary among the species, depending on their distribution and behavior. The value of their meat, eggs, shell or other parts plays an important role in the extent of directed harvest. All sea turtle life stages are vulnerable to human-induced mortality. On nesting beaches, direct exploitation of turtles for meat, eggs, skin or shell, and other products takes place for both commercial markets and local utilization, and to a much lesser degree for traditional ceremonies. Nesting beach and in-water habitat degradation and destruction have occurred due to many factors, including coastal development, dredging, vessel traffic, erosion control, sand mining, vehicular traffic on beaches, and artificial lighting, which repels the adults and disorients the hatchlings. Human alteration of terrestrial habitats can also



change the feeding patterns of natural predators, thereby increasing predation on marine turtle nests and eggs.

Petroleum and other forms of chemical pollution affect turtles throughout their marine and terrestrial habitats. Poisoning, as well as blockage of the gastrointestinal tract by ingested tar balls, has been reported. Low level chemical pollution, possibly causing immunosuppression has been suggested as one factor in the epidemic outbreak of a tumor disease (fibropapilloma) in green turtles. Plastics and other persistent debris discharged into the ocean are also recognized as harmful pollutants in the pelagic environment. Marine turtles such as leatherback turtles actively feed on jellyfish, and plastic bags floating in the water potentially resemble such prey in form, color and texture. Ingested plastics can occlude the gut, preventing or hampering feeding, and causing malnutrition or starvation. Both the entanglement in, and ingestion of, this synthetic debris have been documented (*in* NMFS and USFWS, 1998a-d).

### 3. *Status Summary of Sea Turtle Species*

All listed sea turtle populations affected by the proposed action have been impacted by human-induced factors such as commercial fisheries, direct harvest of turtles, and modification or degradation of the turtle's terrestrial and marine habitats. Nesting beach habitat impacts have resulted in the loss of eggs and hatchlings as well as the deterrence of nesting females resulting in decreased nesting success. In the marine environment, a significant anthropogenic impact is the incidental capture and mortality of subadult and adult sea turtles in various commercial fisheries. Mortality resulting from the effects of marine pollution are important but much less significant. Increased mortality at the egg and early life history stages has impacted the species' ability to maintain or increase its numbers by limiting the number of individuals that survive to sexual maturity. In addition, the mortality of adult females results in the loss of their future reproductive output. The age at sexual maturity of loggerheads may be as high as 35 years, while green turtles may not reach maturity until 30-60 years (*in* Crouse, 1999). Upon reaching maturity, female sea turtles generally lay between 100-130 eggs per clutch, minimally 2-3 clutches per year, every 2-4 years. Thus, in general, a female sea turtle will lay between 200-390 eggs per season over an average of 2-4 years.

The potential for an egg to develop into a hatchling, into a juvenile, and finally into a sexually mature adult sea turtle will vary among species, populations, and the degree of threats faced during each life stage. Females killed prior to their first successful nesting will have contributed nothing to the overall maintenance or improvement of the species' status. Anthropogenic mortality to females (or males, for that matter) prior to the end of their reproductive life results in a serious loss of reproductive potential to the population. While quantitative data do not yet exist to provide a precise understanding of the effects of this loss of reproductive potential, the status and trends of the turtles themselves are the best evidence that sea turtle populations cannot withstand current mortality rates. In the face of current levels of mortality and extent of habitat degradation, nesting assemblages of green, leatherback, and loggerhead turtles have declined to levels that place them at a very high risk of extinction within the foreseeable future. Of the sea turtles considered in this Opinion, only olive ridley turtle nesting assemblages seem to be somewhat stable or increasing slightly.